The long-term impacts of an aerial 1080 application on non-target forest species

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by

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Abstract

The control of introduced mammalian predators in New Zealand forests is crucial for the protection of native species and essential ecosystem services. Possum control in the form of aerial 1080 applications is conducted by TbFree New Zealand to prevent the spread of bovine tuberculosis, and often has the added conservation benefit of temporarily reducing levels of other mammalian predators such as rodents and mustelids. However, native non-target species such as birds and weta can also be at risk of direct and secondary poisoning following 1080 applications, as well as increased predation risk through mesopredator release. To determine whether the benefits of 1080 applications outweigh the risks to non-target native species, both short and long-term monitoring of populations following aerial 1080 applications is needed.

For this study, two forest regions in the South Island were selected for pre- and post-treatment monitoring of non-target species following an aerial 1080 application for possum control. Each region contained a treatment site and a paired non-treatment site. Relative indices of possums, rodents and other mammalian predators were obtained using tracking tunnels and chew cards, indices of birds were obtained using five-minute bird counts, and indices of tree weta were obtained using tracking tunnels and artificial shelters. Monitoring was conducted before the aerial 1080 was applied in August 2012, and over the following 2012/13 and 2013/14 summer seasons.

The aerial 1080 applications were successful at reducing possums to undetectable levels at both treatment sites for the two seasons following treatment. Mice were significantly reduced at one treatment site relative to the paired non-treatment site immediately following the 1080 operation, but had increased to pre-1080 levels by the second post-treatment monitoring season. Rats were detected at low levels, and showed no response to the treatment. Mustelids were not detected at either region throughout the monitoring period.

No native species showed a decline in a treatment site that was not matched in the non-treatment site. Chaffinches significantly declined at both treatment sites relative to non-treatment sites, likely due to an indirect delayed effect such as competition for food resources. Tomtits showed a positive response to the treatment, significantly increasing in both treatment sites over the post-treatment monitoring periods. Tree weta showed no significant decline in response to the treatment. The reduction of possums to low levels, and the maintenance of possum control with ongoing 1080 operations, is likely to continue to provide an overall net benefit to native non-target species.

Chapter 1. Introduction

1.1 Introduced mammalian predators in New Zealand

One of the most important causes of global biodiversity loss is the invasion of ecosystems by exotic species (Courchamp et al. 2003). Since the introduction of rodents, mustelids, possums and feral cats during the colonisation of New Zealand, over 40% of endemic bird species have become locally or globally extinct (Clout 2001, Innes et al. 2010). Like other oceanic islands where native fauna have evolved in the absence of mammalian predators, many bird species on New Zealand's mainland still show a lack of anti-predator recognition behaviour, and are continuing to decline in the presence of introduced mammalian predators (Blackburn et al. 2004, Innes et al. 2010, Jamieson & Ludwig 2012). As well as birds, many species of native plants and invertebrates are also threatened or declining due to predation from introduced mammalian pests, resulting in ecosystem wide impacts to New Zealand's forests.

Of the 35 mammal species introduced to New Zealand, the worst impacts have come from the browsing and predatory behaviour of possums, rodents and mustelids throughout the New Zealand mainland (Saunders & Norton 2001). The Australian brushtail possum (*Trichosurus vulpecula*) was introduced to New Zealand in the 1840's to establish an export fur trade, and its detrimental effects on native ecosystems were not recognized until nearly a century later (Payton 2000). Possums are opportunistic herbivores, feeding on leaves, buds, flowers, fruits and invertebrates, as well as native birds and their eggs (Cowan 2005). Selective feeding by possums on preferred plant species can lead to the defoliation and complete canopy collapse of a forest community within 15-20 years. This selective browsing can result in gradual changes to forest composition through the rapid mortality of vulnerable and preferred species, and is the major cause of the decline of native trees such as pohutukawa (*Metrosideros excelsa*), rewarewa (*Knightia excelsa*), tawa (*Beilschmiedia tawa*) and rata (Metrosideros spp.), as well as mistletoe (*Peraxilla colensoi*, *P. tetrapetala*, *Alepis flavida*, *Ileostylus micranthus*, *Tupeia antarctica*) and fuchsia (Fuchsia spp.) species (Payton 2000, Nugent et al. 2010).

Introduced ship rats (*Rattus rattus*) are found in native and exotic forests throughout the New Zealand mainland, and are most abundant in low elevation mixed podocarp-broadleaf forests (Innes 2005). Ship rats eat seeds, invertebrates, fruits, lizards, and birds and their eggs (Innes et al. 1995, Innes 2005). They are responsible for a large percentage of predation events at the nests of both small and large forest birds (O'Donnell & Phillipson 1996). Several studies have

found that native weta (Orthoptera) make up a significant portion of the diet of ship rats, often being their most consumed prey item (Innes 2005, Ruscoe et al 2012). Both possums and rats are capable of limiting food supplies for a number of bird species due to their consumption of invertebrates, flowers, fruit and leaves (Innes et al. 2010).

Three mustelid carnivores, stoats (*Mustela ermina*), weasels (*M. nivalis*) and ferrets (*M. furo*), were introduced to the New Zealand mainland. Of these, stoats present the greatest predation risk to native species. Stoats were introduced to New Zealand in the 1880's to control rabbits, and are now distributed throughout the mainland. Common prey items for stoats are birds, mice, rats, possums, rabbits, and invertebrates such as weta (King & Murphy 2005). Stoat populations are known to fluctuate in response to the masting cycles of beech forests, when the variable flowering and seeding of beech trees can result in larger than usual seedfall in some years (Wardle 1984). The increase in food supply results in an increase of rodent numbers, allowing stoat populations to increase and breed at numbers far higher than normal (Murphy & Dowding 1995, Kelly et al. 2005). Stoat predation is considered to be one of the main causes of decline in hole nesting bird species such as kaka (*Nestor meridionalis)*, mohua (*Mohoua ochrocephala*) and yellow-crown parakeets (*Cyanoramphus auriceps*), which are especially vulnerable while nesting (O'Donnell & Phillipson 1996, Wilson et al. 1998). During beech masting years, the predation rate on native birds can increase considerably (O'Donnell & Phillipson 1996, Dilks et al. 2003).

1.2 Ecosystem impacts of mammalian predation

The decline of bird species can result in the decline of important ecosystem processes such as seed dispersal and pollination, threatening the plant species that depend on mutualistic relationships with birds (Sekercioglu et al. 2004, Kelly et al. 2006). Native species that provided these ecosystem services in pre-European New Zealand, and that are now extinct or in decline, include moa (Dinornithiformes), kokako (*Callaeas cinerea*), hihi (or stitchbird, *Notiomystis cincta*), short-tailed bats (Mystacinidae), and lizards (Newstrom & Robertson 2005, Kelly et al. 2006). Currently, pollination and seed dispersal in New Zealand is primarily dependent on four native bird species; tui (*Prosthemadera novaseelandiae*), bellbird (*Anthornis melanura*), silvereye (*Zosterops lateralis*) and stitchbird (Kelly et. al 2006). The stitchbird is now absent from the mainland, bellbirds are absent from the northern North Island, and tui are rarely found on the east of the South Island (Montgomery et al. 2001). As a consequence, pollen limitation is now widespread in New Zealand for many species with ornithophilous flowers (Kelly et al. 2010), and some species (such as *Rhabdothamnus solandri*) are pollen limited on the mainland but not on offshore predator-free islands with

more intact avifaunas (Anderson et al. 2011). Without continued control of introduced mammalian predators on the mainland, ecosystem services may continue to decline.

1.3 Options for control of mammalian pests

The eradication of introduced predators from offshore islands has shown that recovery of native flora and fauna is possible, with new populations of vulnerable species established on predator-free islands through translocation (Saunders & Norton 2001). However, offshore islands are not large enough to maintain all of New Zealand's remaining biodiversity, and conservation focus has recently been shifted to creating mainland 'islands', and using extensive methods of predator control to keep areas free of predators (Norton 2009). Various methods for controlling pests are available. Non-toxin techniques include trapping, shooting, and the use of predator-proof fences. Trapping and shooting are both labour intensive, and many remote areas of forest are not accessible by foot. In recent decades, aerial and ground poisoning using toxic baits has been the main method of large-scale pest control in New Zealand. Poisoning is a rapid and cost-effective technique that can be used over large areas of inaccessible forest, and can kill multiple pest species through either direct or secondary poisoning (Morgan & Hickling 2000). There are six poisons currently registered for possum control in New Zealand: 1080 (sodium fluoroacetate), cyanide, brodifacoum, cholecalciferol (vitamin D), pindone, and phosphorous. Of these, 1080 is the poison used most extensively for the control of both possums and other mammalian pests (Eason et al. 2011).

1080 as a mammalian pest control

Sodium fluoroacetate, commonly referred to as 1080, was developed as a vertebrate pesticide in the 1940's. Fluoroacetate is a toxic component of plants found in Australia, South America and South Africa, and has been used for mammalian pest control in New Zealand since the 1950's. Currently New Zealand is the most extensive user of 1080, followed by Australia – both countries have extensive problems with introduced mammals preying on endemic species (Eason et al. 2011). The toxin acts by interfering with cellular energy production, and lethal doses can kill a mammal within 6-48 hours of bait consumption (Eason et al. 2011). The poison is distributed in baits that typically consist of carrot or cereal-based pellets, laced with a concentration of 1080. The baits can be applied to a desired treatment area either aerially or on the ground. Possums and rodents can be killed directly by consuming a lethal dose of the bait, and mustelids can be killed through secondary poisoning (i.e., consuming a poisoned rodent). There are four groups that conduct aerial 1080 applications: TbFree New

Zealand (known as the Animal Health Board until July 2013), the Department of Conservation, regional councils, and private land owners (Table 1.1).

Table 1.1. Objectives, scale and frequency of aerial 1080 applications by the main users. Information obtained from Innes & Barker 1999, Brown & Urlich 2005 and a 2012 annual report from the Environmental Protection Authority.

	Objective	**Scale of aerial 1080 applications in 2012**	**Frequency**
TbFree New Zealand	Eradication of Bovine tuberculosis	280,000 ha	Repeat 3-5 years
Department of Conservation	Biodiversity protection	136,000 ha	Varies 3-7 years
Regional councils	Biodiversity and production protection	5,200 ha	Variable
Private land owners	Farming profitability, rabbit eradication	12,000 ha in Canterbury and Otago, nationally unknown	Frequent

1.4 The role of Bovine tuberculosis in the application of 1080

Bovine tuberculosis (Tb), an infectious disease caused by *Mycobacterium bovis*, is an introduced disease in New Zealand that can infect farmed cattle and deer, and which if present at high incidence could threaten New Zealand's trade in dairy, beef and venison products (Coleman & Cooke 2001). Four introduced and wild mammal species (possums, red deer, ferrets and pigs) are frequently infected in New Zealand, but the possum is regarded as the only true maintenance host, and the transmission of Tb from possums to livestock is considered the greatest barrier to the eradication of Tb from New Zealand livestock (Coleman & Caley 2000, Nugent 2011).

TbFree New Zealand is a government-industry partnership. The organisation's primary role is to manage and implement the National Pest Management Plan for Bovine Tb, under the Biosecurity Act 1993. Their aim is to eradicate Bovine Tb from New Zealand by managing the movement of the disease in cattle and deer herds, and conducting possum control operations using a combination of ground control methods and aerial 1080 applications. In 2012, TbFree New Zealand treated approximately four percent (280,000 hectares) of its total treated area with aerial applications of 1080. Treated areas are typically re-treated in three to five-year cycles, and often include areas administered by the Department of Conservation (Brown & Urlich 2005). The operations, although aimed at Tb eradication, also

contribute greatly to national biodiversity protection through the reduction of possums as well as non-targeted mammalian pests such as rodents and mustelids.

1.5 Evaluating 1080: science and public concerns

Public perceptions

The use of aerial 1080 as a pest control method has been controversial with the New Zealand public since its use became widespread. During a national survey conducted in 2007, members of the general public were asked "Do you support or oppose the use of 1080 poison to control the impact of possums on native forests and birds, and to control the spread of tuberculosis?" 45% of respondents supported use, and 43% opposed (Green & Rohan 2012). Public submissions made to the ERMA (Environmental Risk Management Authority) in 2007 showed that public perceptions of the risk of 1080 are largely based around concerns for people's health (such as contamination of waterways), contamination of forests, and the by-kill or secondary poisoning of native bird species, deer, pigs, dogs, and other domesticated or farm animals (Green & Rohan 2012).

Governmental reports in response to public concern

Although 1080 had been previously cleared for use in New Zealand, the Environmental Risk Management Authority (ERMA) reassessed the use of 1080 in 2007, in response to applications made by the Department of Conservation and TbFree New Zealand seeking approval for the continued use of 1080 as a pest control tool. The ERMA concluded that 1080 should continue to be used for pest control, as there is currently no practical alternative for forest and agriculture preservation. However, they also recommended the monitoring of future aerial operations, along with continued research into alternative pest control techniques.

Largely in response to public concern surrounding the use of 1080, a 2011 report was issued by Dr. Jan Wright, the Parliamentary Commissioner for the Environment, to provide an independent assessment of the use of 1080 in New Zealand. The report reached the same conclusions as the ERMA - that the toxin poses no discernable threat to human safety. Dr. Wright recommended that the use of aerial 1080 operations for the protection of forests should not only continue but also increase, as alternative control tools lack the same advantages.

Scientific research on 1080 and its environmental effects

Studies have shown that although 1080 is highly water soluble, baits that fall into streams during aerial 1080 application are biodegraded by aquatic plants and organisms within 2-6 days (Parfitt et al. 1994). The concentrations are also greatly diluted, and it is unlikely that any significant contamination of waterways would occur following a 1080 application (Eason et al. 2011). Under warm and wet conditions, 1080 residues will typically biodegrade or become diluted by rainfall within 1-4 weeks of application (King et al. 1994). Residues of 1080 remain in possum and rodent carcasses for up to one week following poisoning (Eason et al. 1993). Plants may absorb 1080 that has leached into the soil, but the toxin is degraded by the plants within one to two months (Eason et al. 2011).

Effects on non-target species

Without proper bait preparation and application, non-target species including native birds are susceptible to 1080 poisoning (Eason et al. 2011). Many deaths of non-target bird species were reported during 1080-poisoning operations conducted in the 1970's. The 1080 poisoning operations at this time were conducted using carrot-based baits that were non-dyed and contained a raspberry-flavoured lure, and a range of bait fragment sizes from large to small were distributed (Spurr 2000, Eason et al. 2011). These factors all contributed to the consumption of the baits by birds, which led to a significant change in the way 1080 baits are prepared and distributed. Baits are now dyed green to make them less attractive to birds (Caithness & Williams 1971), carrot baits are screened through a fine grid to remove fragments, and cereal-based baits with a cinnamon oil deterrent have become more common as they are thought to kill fewer birds (Spurr 1991). Since these changes took place in 1977, fewer birds have been recorded dead as a result of aerial 1080 applications (Spurr & Powlesland 1997). Recent studies of long-term impacts of aerial 1080 operations on bird populations (O'Donnell & Hoare 2012, Greene et al. 2013) have found no negative impacts of 1080, and suggest that some species can show significant increases in response to the reduction of predator levels.

Invertebrates, including native weta species, are also known to consume 1080 baits, with low concentrations of 1080 detected in live weta in the forest up to four weeks after 1080 operations (Eason et al. 1993). Monitoring studies suggest that 1080 operations are not likely to have a negative effect on the populations of weta and other invertebrates, as the proportion of individuals feeding on the baits would be small compared to the total numbers of invertebrates in the area (Spurr & Drew 1999, Spurr & Berben 2004, Powlesland et al. 2005). Weta populations may also benefit from a release of mammalian predation following a 1080-poisoning operation (Ruscoe et al. 2012).

1.6 Thesis outline and experimental design

The aim of this thesis is to present and discuss results from the first two years of a long-term monitoring project on changes in native forests in relation to possum control with aerial-1080. The experimental design of the project involves measuring an index of population density of mammalian predators, non-target bird species and tree weta in treatment and non-treatment areas before and after the aerial application of 1080 in the treatment areas. This design assumes that in the absence of treatment, the trends of non-target species would change in a similar way in both areas over time. Two different study areas (each containing a treatment site and paired non-treatment site) were chosen. Aerial 1080 was applied to the treatment sites in August 2012 after pre-treatment monitoring in June-July, and post-treatment monitoring was conducted during November-January 2012/13 and December-January 2013/14.

Monitoring methods included chew cards and tracking tunnels for mammals, five-minute bird counts and automatic bird recorders for birds, and the use of tracking tunnels and artificial shelters for tree weta. These techniques all provide a relative measure of population density, and were conducted along permanently placed transect lines (See Chapter 2.2. for maps).

Study sites

Two regions in the South Island of New Zealand were chosen as study areas, as both contained a suitable treatment site and a matched non-treatment site with similar forest composition. During 2012, both treatment sites were treated with an aerial 1080 application conducted by TbFree New Zealand. The objective of both operations was to reduce the possum population to below 2% RTCI (residual trap catch index).

Rolleston Range

Rolleston Range is located near Lake Coleridge in the Canterbury region, and the forest is dominated by mountain beech (*Nothofagus solandri*). The elevation of the treatment and non-treatment sites ranges from 620-1220m. The treatment site (23,337 ha, Fig, 1.1) was treated with pre-feed non-toxic cereal baits on 18-19 July 2012, and 1080-laced baits were aerially distributed on 25-27 July 2012. The baits were 0.15% 1080 cereal-based pellets, containing 1.5-2.0g of 1080/kg. Cinnamon lures and deer repellent were added to the baits. This area had never been treated with 1080 prior to 2012. The paired non-treatment site, chosen for similar forest composition and elevation to the treatment area, is located near the Harper River, 22km to the east (for site maps, see Chapter 2.2).

Alexander Range

Alexander Range, near Otira on the West Coast, is composed of lowland mixed native forest, and the elevation of the treatment and non-treatment sites ranges from 140-860m. The treatment site (7081 ha, Fig. 1.2) was treated with pre-feed non-toxic cereal baits on 21 July 2012, and 1080-laced baits were aerially distributed on 10 August 2012. The bait type was the same as for Rolleston Range, except deer repellent was not added. Several earlier 1080 applications have occurred at this site. The paired non-treatment site is located near the Upper Taipo River, 17km to the south (for site maps, see Chapter 2.2).

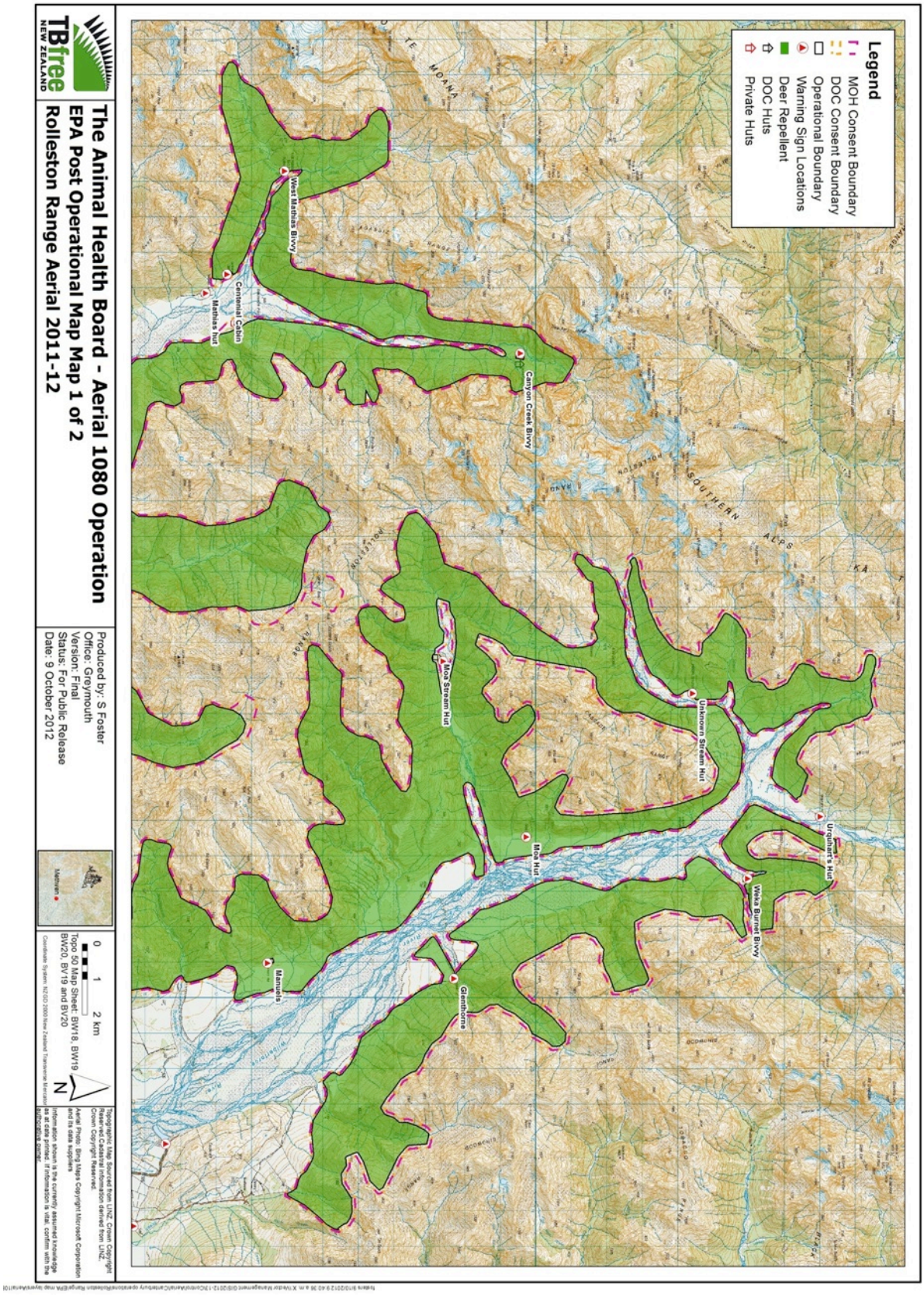

Figure 1.1. Operational map showing the areas of Rolleston Range treated with aerial 1080 in 2012.

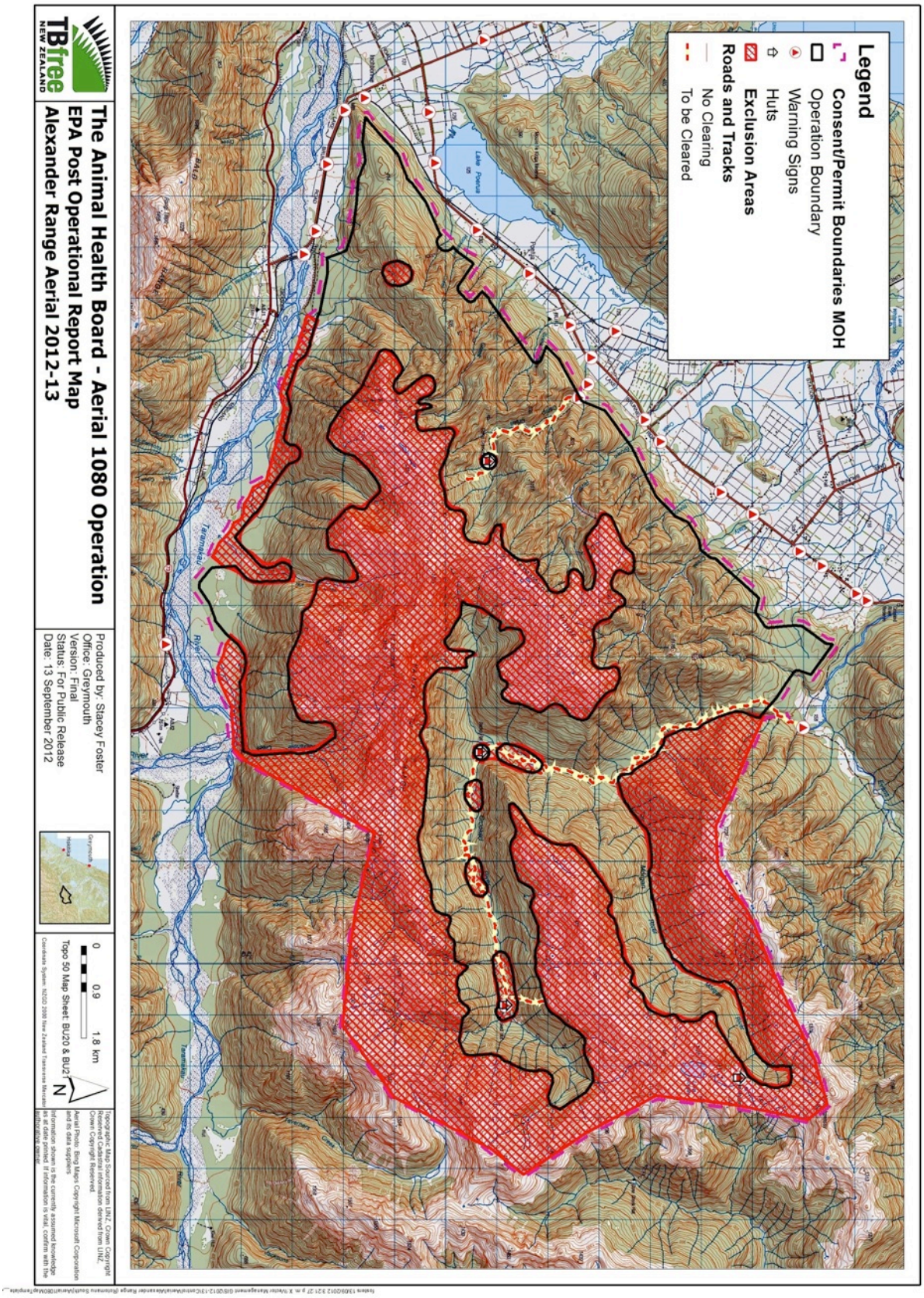

Figure 1.2. Operational map showing the areas of Alexander Range treated with aerial 1080 in 2012.

2. Mammal monitoring

2.1 Introduction

Large-scale possum control operations have been a main focus of New Zealand conservation management for the last four decades (Nugent et al 2010). Trapping, shooting, chemical repellants and physical barriers have all been used in the management of possums, but poisoning is the most extensively used large-scale technique on conservation land in New Zealand (Montague & Warburton 2000).

Controlling for one pest species when other pest species are also present poses risks in the form of unwanted responses by the pests that are not controlled, which can reduce the overall net benefit of pest control (Sweetapple & Nugent 2007). Aerial 1080 operations primarily target possum populations, but populations of other mammalian pest species living in New Zealand forests can also be affected, both directly and indirectly. The baits can kill large proportions of ship rats through primary poisoning, and variable proportions of stoats through secondary poisoning. Populations of ship rats and house mice have been known to increase in response to reduced possum densities some months after poisoning (Sweetapple et al. 2006). As possums, ship rats and stoats are all significant predators of native bird and weta species, understanding both short and long-term population responses of mammalian pests to 1080 poisoning is important for conservation management.

Kill rates for possums using aerial 1080 typically range between 75 and 100 percent, although recent developments to improve the accuracy of aerial sowing has put the usual kill rate above 90 percent (Morgan & Hickling 2000). Following a successful large-scale control operation, possum populations can take between 6 to 10 years to recover to pre-treatment levels (Clout & Barlow 1982, Nugent et al. 2010). When possum control for conservation is the primary goal, aerial 1080 applications are typically repeated every 5 to 10 years (Nugent et al. 2010). When the primary goal is to eradicate Tb, 1080 applications can be repeated as often as every 3 to 5 years (Brown & Urlich 2005).

Most aerial 1080 operations reduce ship rat abundance by at least 90%, but recovery to pre-treatment levels is much quicker than that of possum populations (Powlesland et al. 2000). There is evidence to suggest that when both possum and rat densities are reduced by 1080, rat density can recover to pre-treatment levels or higher within two to six months (Innes et al. 1995). Sweetapple et al. (2006) monitored indices of possum, rodent and stoat abundance for

three years following a single aerial 1080 application. For at least one breeding season during the three years after possum control, ship rat abundance was significantly higher in treated blocks. This may be due to reduced competition for food resources, as ship rats and possums share a preference for high-energy food such as nuts, fruit and invertebrates (King 2005, Ruscoe et al. 2011). In areas where possum control is the main goal, and control operations are not carried out on an annual cycle, the possible increase in rat abundance could have important negative impacts on the predation of native species.

Responses of mouse populations to 1080 poisoning operations are less well known. In some studies mouse populations have been found to increase dramatically within 2-6 months of control, but only if the operation reduced rat abundance to low levels, or in areas such as high altitude beech forests were rats are normally absent (Innes et al. 1995, Ruscoe et al. 2011).

Stoats are affected indirectly by secondary 1080 poisoning. Rodents are an important vector of the toxin to stoats and other predators such as feral cats, which don't eat the baits directly but can be killed by consuming a poisoned rodent (Gillies & Pierce 1999, King 2005). Even at low prey density, a large proportion of stoats can be killed following 1080 operations (Gillies & Pierce 1999). Due to their high potential productivity and opportunistic diets, stoats are thought to be able to recover from 1080 operations during the next spring breeding season, providing rodent abundance also increases (King 2005, Sweetapple et al. 2006).

Monitoring techniques

Using tracking tunnels to monitor the abundance of small mammals in New Zealand was first described by King & Edgar (1977), and is commonly used by conservation managers and researchers at mainland sites throughout New Zealand. It is a non-destructive sampling technique with low costs, and is thought to be more sensitive than snap traps for detecting the presence of rodents at low abundances (Gillies & Williams 2013). Results provide an index of relative abundance rather than a direct measure of population density, and the method is best at providing comparisons between similar habitat areas over time.

Tracking tunnels detect small mammals such as rats, mice and stoats, but possums are usually unable to fit into the tunnels. Non-destructive monitoring for possums in New Zealand is often carried out using a wax-tag or card interference method. The proportion of cards or tags interfered with (usually "chewed") provides an index of relative abundance (Spurr 1995). Ensuring that monitoring stations are at least 40 m apart reduces the chance of individual possums leaving signs at more than one station (Bamford 1970, Jane 1981). Wax-tags are not baited, and simply attract animals by curiosity and scent. Possums, rats and mice

leave distinctive bite marks, but the marks can be obscured or the entire block destroyed by later visitors (Sweetapple & Nugent 2011). Chew-track-cards are baited with food such as peanut butter or alfalfa, and distinctive bite marks of possums, mice and rats can be recorded.

Objectives

This chapter presents the results of mammal monitoring. The purpose of the monitoring was to determine how mammalian predators responded to a single aerial 1080 drop, using tracking tunnels and chew cards. The monitoring was carried out over three seasons (one pre-treatment and two post-treatment) at study sites at the Rolleston and Alexander regions. Relative indices of mammal populations were compared between treatment and paired non-treatment sites and over time.

Predictions of the response of mammalian predators to the 1080 operation were as follows:

1) Mammalian pest density in the non-treatment sites will remain stable throughout the study.

2) Possum density will decrease to very low levels in treatment sites, and remain low during the second season post-treatment.

3) Rodent density will decrease to very low levels in treatment sites, but is likely to increase by the second season post-treatment.

4) Stoat density will decrease in treatment sites, and an increase in density in the second season post-treatment may be dependent on the recovery rate of rodents.

2.2 Methods

Pre- and post-treatment measurement of mammalian predators was carried out using chew cards and tracking tunnels. Animal Health Board contractors and a team from the University of Canterbury completed two sets of winter 2012 pre-treatment monitoring. A team from the University of Canterbury (Josh Van Vianen, Archie McFarlane and myself during summer 2012/13) completed the post-treatment monitoring. All mammal monitoring at each site occurred along or near established tracks with permanently marked stations (Fig. 2.1, 2.2).

2.2.1 Chew cards

Chew card monitoring was completed once per site, per season. Standard 'Connovation' pre-baited chew cards were installed every 50m along established lines (Fig. 2.1), with a total of 50 cards per site spaced in five lines of 10. The cards were exposed for three nights and scored by Josh Van Vianen and Archie MacFarlane from the University of Canterbury. Card monitoring was repeated during summer 2012/13 at the Harper study site, as the cards installed in December 2012 were only exposed for two nights due to adverse weather conditions. Summer 2013/14 monitoring was repeated at the Wilberforce site after the first set of cards in October 2013 were exposed for 14 nights, also due to weather conditions (Table 2.1). These non-standard sets have not been included in analysis.

Table 2.1. Dates of chew card monitoring at the Rolleston and Alexander Ranges over three seasons. Parentheses mark sets not exposed for three nights, and not used in analysis.

	Rolleston Range		**Alexander Range**	
	Wilberforce (T)	Harper (NT)	Alexander (T)	Taipo (NT)
Winter 2012	30 June-3 July	19-22 June	1-4 July	8-11 Aug
Summer 2012/13	27-30 Nov	(4-6 Dec) 20-23 Jan	18-21 Dec	11-14 Dec
Summer 2013/14	(1-14 Oct) 21-24 Jan	14-17 Jan	3-6 Dec	10-13 Dec

2.2.2 Tracking tunnels

During winter 2012, two sets of tracking tunnels were run – the 'A' set, installed by Animal Health Board contractors, and the 'B' set, installed by the University of Canterbury. The 'A' set included 50 tunnels per site, placed along five lines of 10 per line and spaced 50m apart (Fig. 2.2). These tunnels were left out for three nights, and baited with 'Erayze' polymer rabbit bait to attract stoats and rodents. The 'B' set of tunnels included 40 tunnels per site in 4 lines of 10 per line, spaced at 100m and baited with peanut butter, and exposed for two nights. From summer 2012/2013 onwards only the 'A' set of tunnel locations were used. As there was no sign of stoats during winter 2012, subsequent tunnels were baited with peanut butter

and exposed for one night, following the recommended Department of Conservation protocol for rodents (Gillies & Williams 2013).

For comparisons between seasons and sites, the 'B' set tunnels that were exposed for two nights were used for winter 2012, as this method most closely matched the method used post-treatment (Table 2.2a). In winter 2012, weather conditions meant that some additional cards from the 'B' set were exposed for 3 and 7 nights at Wilberforce. (Table 2.2b).

All pre-treatment data were separately analysed as a comparison of method, to test for the effect of bait type (peanut butter or rabbit), and the number of nights exposed (2-7). All tunnel cards were scored by Josh Van Vianen and Archie McFarlane at the University of Canterbury.

Table 2.2. Dates of tracking tunnel monitoring at the Rolleston and Alexander ranges over three seasons. (a) standard sets used in full analysis.

	Rolleston		**Alexander**	
	Wilberforce (T)	Harper (NT)	Alexander (T)	Taipo (NT)
Winter 2012	28-30 June	28-30 June	4-6 July	26-28 July
Summer 2012/13	29-30 Nov	4-6 Dec, 22-23 Jan	18-19 Dec	13-14 Dec
Summer 2013/14	1-2 Oct 23-24 Jan	14-15 Jan	10-11 Dec	3-4 Dec

(b) Additional sets from winter 2012, run for various lengths of exposure.

	Rolleston		**Alexander**	
Winter 2012	Wilberforce (T)	Harper (NT)	Alexander (T)	Taipo (NT)
A set	30 June-3 July	19-22 June	7-10 July	8-11 Aug
B set	16-19, 22-28, 28-30 June	28-30 June, 20-22 July	4-6 July	26-28 July

2.2.3 Statistical analysis

Chew card and tracking tunnel data were analysed using a repeated measures binomial GLM in R. Analysis was done separately for Rolleston Range and Alexander Range. Counts were grouped at the transect level (10 tunnels/cards per transect) to allow for the nested nature of the data (multiple cards and transects). The key test was for a significant 'season x treatment' interaction, which shows a possible effect of 1080 as counts at the treatment site have changed over time in a different way to the non-treatment site.

Where the season x treatment interaction was significant, post-hoc multiple means comparisons were run using Tukey contrasts (R package 'multcomp' and command 'ghlt'). For the Tukey tests, another GLM was run using 'site/date' as a predictor to compare all site/season means (variables were grouped as Harper2012, Harper2013, Wilberforce2012 etc).

The large number of zeros in the dataset posed problems for analysis. Some site/season groups contained all zeros due to there being no sign of a mammal species. This caused a problem called ‘complete separation’ to occur within the model, which resulted in inflated variances for that site/season group. This could not be solved by combining results per transect or across sites, as the number of zeros was still too high. Fitting a zero-inflated negative binomial model or a binomial GLMM with a random term for card/observation failed to solve the problem, as the variances were still inflated.

My solution was to adopt a conservative approach by reducing the difference between variables with low and high numbers. I added one ‘1’ count to groups which contained all zeros (e.g. presence of mice shown by chew cards for Harper in season 3 was changed from 0/50 to 1/50, which reduced the difference between means). This reduced the chance of obtaining a significant result, but eliminated the complete separation error and allowed conservative P values to be obtained.

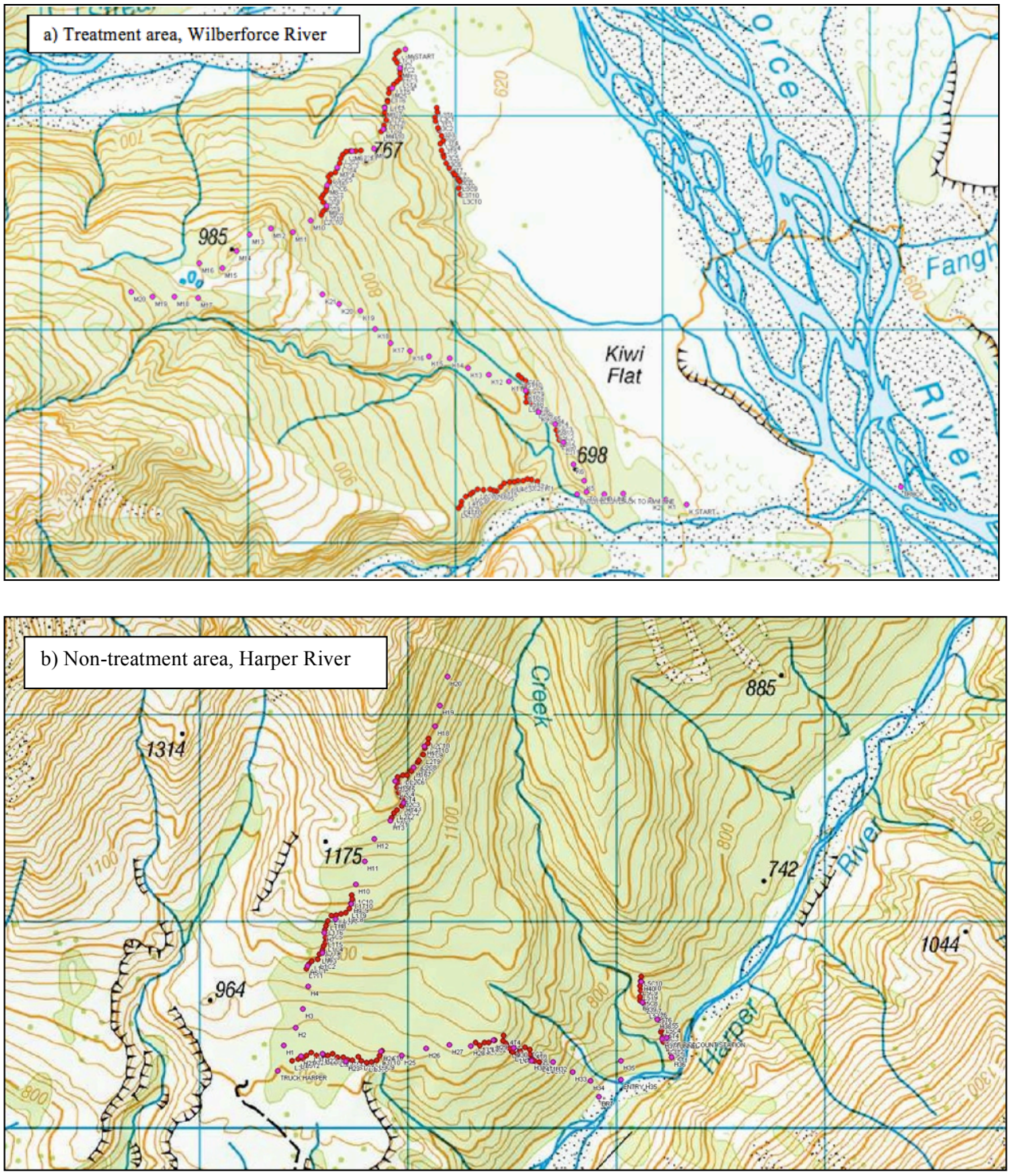

Figure 2.1. Location of permanent bird counting stations (pink, usually alternate stations used) and tracking tunnel/chew card lines (red) in the Rolleston Range region. (a) Treatment site, Wilberforce River, (b) Non-treatment site, Harper River.

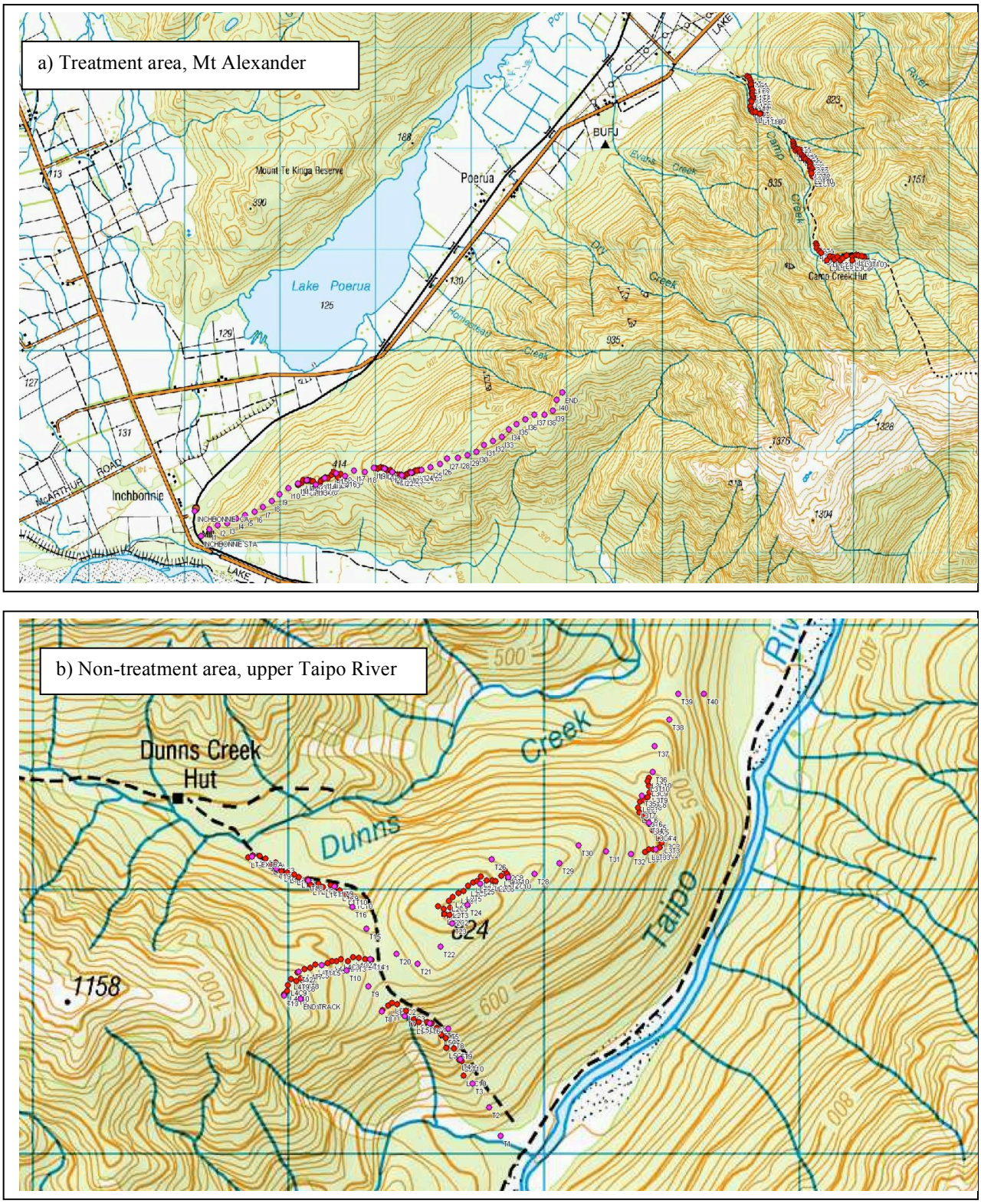

Figure 2.2. Location of permanent bird counting stations (pink, usually alternate stations used) and tracking tunnel/chew card lines (red) in the Alexander Range area. (a) Treatment site, Mt Alexander, (b) Non-treatment site, upper Taipo river.

2.3 Results

Possums, ship rats and house mice were the most common mammalian pest species recorded during monitoring. Rats were found at higher levels at Alexander Range, whereas mice were the most common rodents at Rolleston Range. No stoat tracks were recorded in either region.

2.3.1 Tracking tunnel method comparison

During winter 2012, two sets of tracking tunnels were run with different combinations of baits and exposure nights (Fig. 2.3, 2.4). Since no stoats were recorded in the rabbit-baited tunnels, the method for the following seasons was changed to peanut butter bait and a one night exposure, which is recommended for rodents. GLM analysis of the different bait/exposure nights used during winter 2012 showed that there was no signficiant method effect for mice (p=0.11), rats (p=0.87) or possums (p=0.80) at either region. Tracking tunnels are not designed for the monitoring of possums, and possum marks were found only occasionally.

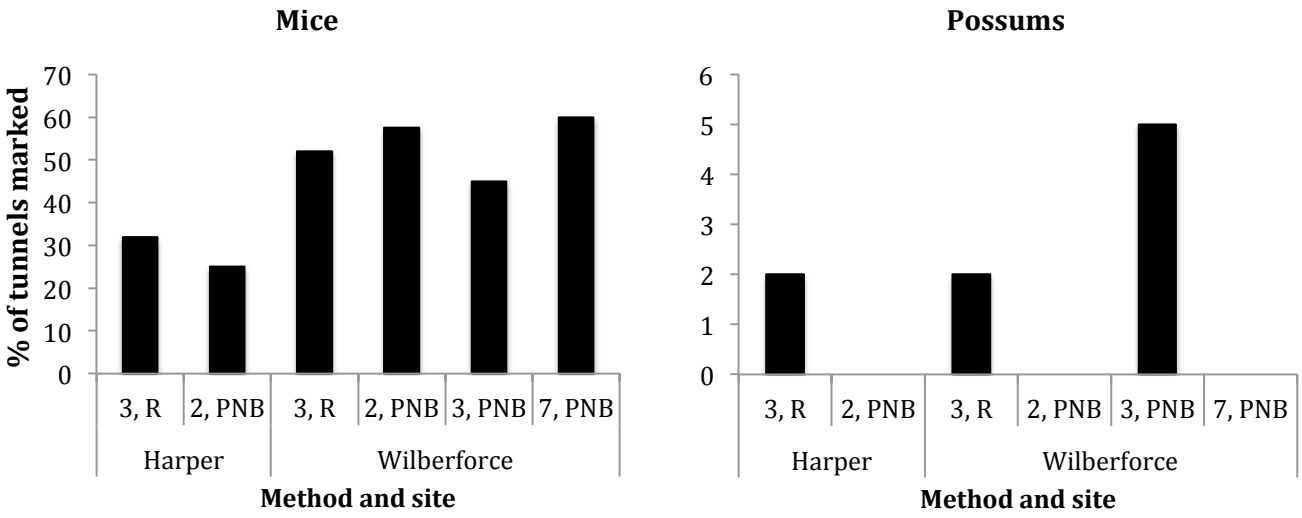

Figure 2.3. The percentage of tracking tunnels marked by mice (left) and possums (right) at Rolleston Range during winter 2012 for each bait type/exposure night combination. PNB=peanut butter, R=rabbit polymer. Signs of possum were zero for some sites/methods.

The tracking tunnel data using the 'two nights, peanut butter' method from winter 2012 was used for analysis with data from the two summer seasons, which both used a 'one night, peanut butter' method. This means that the pre- vs. post-treatment comparisons are also confounded with a shift from two nights exposure to one night. However as there was little difference between all methods used during winter 2012, it is likely that the means derived from the one and two night peanut butter methods are similar.

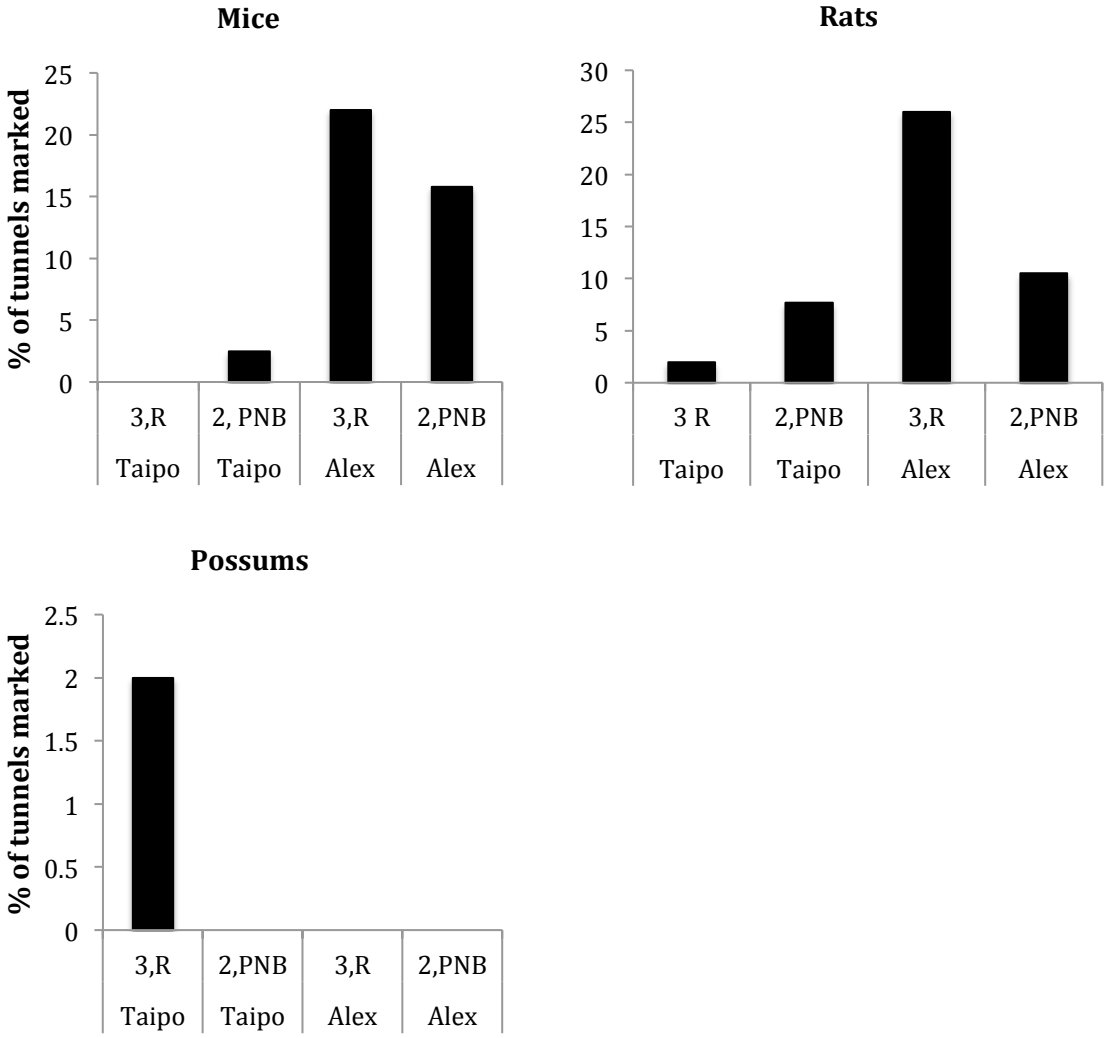

Figure 2.4. The percentage of tracking tunnels marked by mice (above left), rats (above right) and possums (below left) at Alexander Range during winter 2012 for each bait type/exposure night combination. PNB=peanut butter, R=rabbit polymer. Signs of possums and mice were zero for some sites/methods.

2.3.2 Rolleston Range

The two most common mammal species found at Rolleston Range were possums and mice (Table 2.3, 2.4). Very low levels of cats and rats were detected in the tracking tunnels (Table 2.3). GLM analyses showed that the 'season x treatment' effect was significant for both possums and mice (Table 2.5), showing different response patterns at the two sites over time. 'Season' effects were significant for possum (chew cards and tracking tunnels) and mice (chew cards only), and 'site' effects were significant for possums and mice (both chew cards and tracking tunnels).

Table 2.3. Tracking tunnel totals of mammals at Harper (non-treatment) and Wilberforce (treatment) over three seasons. "Nights" is number of nights exposed, and peanut butter bait was used."Total" is the number of tunnels exposed; a tunnel may have signs for several different animals. "Set" is the exact location of tunnels within the site.

Season	**Winter 2012**	**Summer 12/13**	**Summer 13/14**	**Winter 2012**	**Summer 12/13**	**Summer 13/14**
Treatment	Non-tmt	Non-tmt	Non-tmt	Tmt	Tmt	Tmt
Nights	2	1	1	2	1	1
Set	B	A	A	B	A	A
Possum	0	9	12	0	0	0
Mouse	10	26	0	23	1	38
Rat	0	1	2	0	0	0
Cat	2	0	0	1	0	0
Hedgehog	0	0	0	0	0	0
Blank	25	4	8	17	46	11
Total	40	50	50	40	50	50

Table 2.4. Chew card totals of mammals in the Rolleston region over three seasons, pre and post-treatment. "Total" is the number of cards exposed; a card may have signs for several different animals.

Season	**Winter 2012**	**Summer 12/13**	**Summer 13/14**	**Winter 2012**	**Summer 12/13**	**Summer 13/14**
Treatment	Non-tmt	Non-tmt	Non-tmt	Tmt	Tmt	Tmt
Possum	9	39	47	4	0	1
Mouse	4	2	0	14	1	28
Rat	0	0	0	0	0	0
Blank	37	10	3	32	49	21
Total	50	50	50	50	50	50

Prior to the 1080 treatment, the numbers of chew cards marked by possums were similar in both sites (Fig. 2.4). Possum levels in the non-treatment site significantly increased over both intervals following the 1080 application (Fig. 2.4, Table 2.6), whereas no possums were detected during summer 2012/13 at the treatment site, and there was very low detection (only one marked card) during summer 2013/14 (Table 2.4). A significant 'season' effect for possum chew card detection shows that possum detection was lowest during winter 2012 (Fig. 2.4, Table 2.5).

Some tracking tunnels, which are not designed for possum detection, were marked by possums in the non-treatment site. There was no significant season x treatment effect for these tracking tunnels (Table 2.5) despite a similar pattern to the chew cards being observed, with higher detection at the non-treatment site than the treatment site following the 1080 operation (Fig. 2.4). Low tracking tunnel rates for possums, combined with a small sample size, may have resulted in low power of the GLM to detect an interaction.

Both chew cards and tracking tunnels showed a similar pattern for mice at the treatment site (Fig. 2.4). Mouse levels significantly decreased over the first interval from 57% of tracking tunnels marked to 2%, but increased over the second interval to become higher than pre-treatment levels with a 76% tunnel detection rate (significant season x treatment effect, Table 2.5). Chew card detection rates significantly decreased from 28% in winter 2012 to 2% in summer 2012/13, then significantly increased to 56% in summer 2013/14 (Fig. 2.4).
At the non-treatment site, chew card detection remained low for mice throughout the study. However, tracking tunnels showed a different pattern, with high levels of mice (52%) detected during summer 2012/13. No mice were recorded on either chew cards or tracking tunnels during summer 2013/14. Possum signs on chew cards were extremely high during the summer monitoring periods at the non-treatment site (78-94% of cards marked), and it is possible that this obscured any marks made by mice.

Table 2.5. Summary of significant predictors for Rolleston mammal data in poisson GLMs (P values for the effects listed). For full details of each GLM see Appendix A.

Species	**Season**	**Site**	**Season:treatment**
Mouse, chewcard	**<0.001**	**<0.001**	**0.002**
Mouse, tracking tunnel	0.08	**0.002**	**<0.001**
Possum, chewcard	**<0.001**	**<0.001**	**<0.001**
Possum, tracking tunnel	**0.023**	**<0.001**	**0.311**

Rats were not detected at either site prior to 1080 treatment, and were detected at very low levels (4% tracking rate, Table. 2.3) at the treatment site during the second post-1080 season.

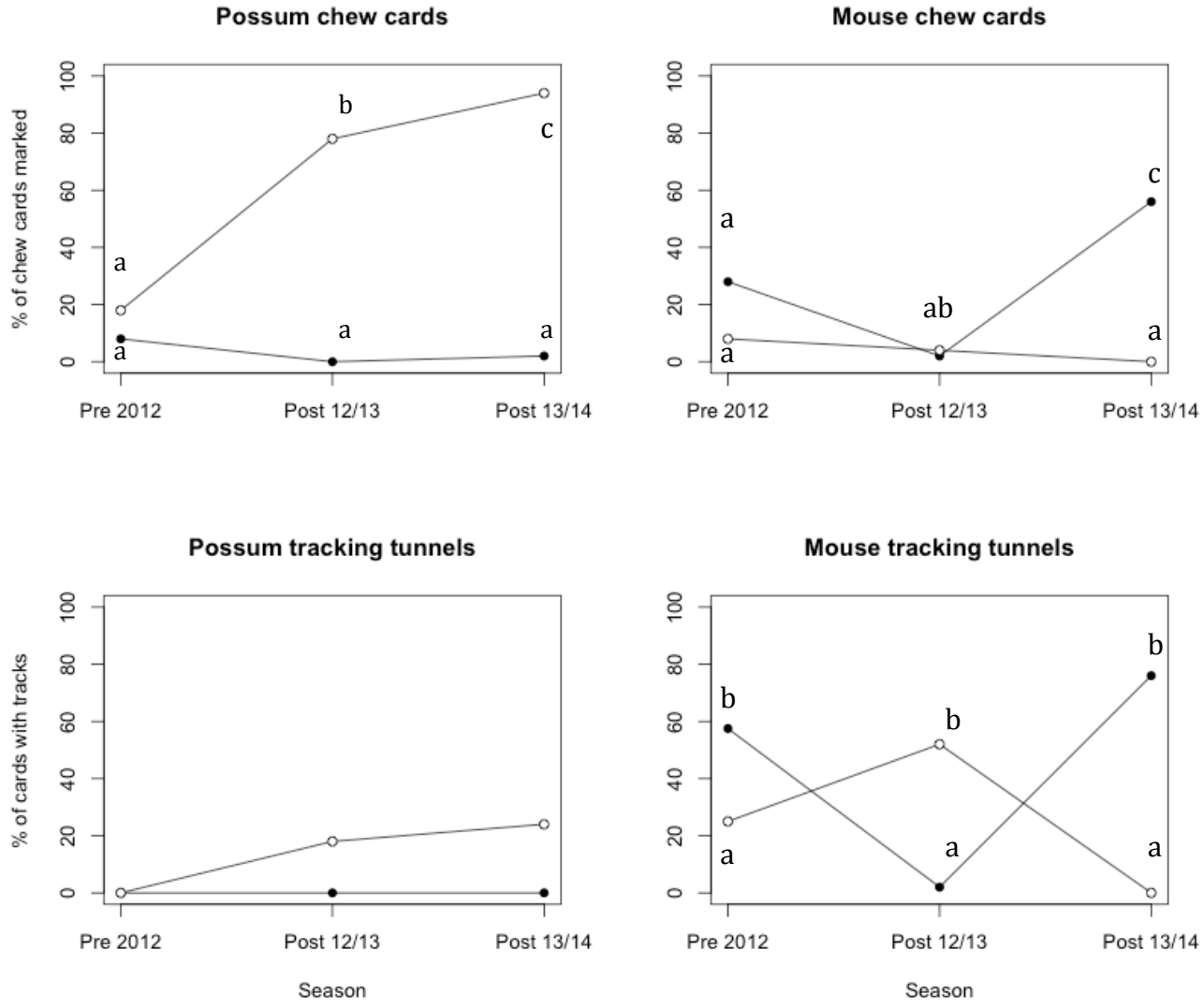

Figure. 2.4. The percentage of marked chew cards and printed tracking tunnels collected during mammal monitoring across three seasons for possums and mice at Rolleston Range. Black circles represent the treatment site, and hollow circles represent the non-treatment site. The letters on the graphs indicate the results of post-hoc tukey analysis, which was performed when a significant 'season x treatment' interaction was present (see tables 2.6a and b); means that are significantly different are represented by different letters.

Table 2.6. Z and P values for post-hoc multiple comparisons of means (Tukey contrasts). 'H' or 'W' refers to site/treatment (Harper or Wilberforce), and seasons are labelled as 1 = pre-treatment, 2 = post-treatment 2012/13, and 3 = post-treatment 2013/14.

a) Tracking tunnel analysis for mice.

	Mouse	
Site & season	*Z* value	*P* value
H1-H2	2.551	0.095
H1-H3	-2.600	0.083
H2-H3	-3.786	<0.001
W1-W2	-3.958	<0.001
W1-W3	1.847	0.401
W2-W3	4.745	<0.001
H1-W1	2.886	<0.05
H2-W2	-3.786	<0.001
H3-W3	4.745	<0.001

b) chew card analysis for mice and possums.

	Mouse		**Possum**	
Site & season	*Z* value	*P* value	*Z* value	*P* value
H1-H2	-0.826	0.957	5.541	<0.001
H1-H3	-1.275	0.776	6.096	<0.001
H2-H3	-0.575	0.991	2.165	0.238
W1-W2	-2.786	0.05	-1.275	0.785
W1-W3	2.792	0.05	-0.826	0.959
W2-W3	3.938	<0.001	0.575	0.992
H1-W1	2.459	0.120	-1.451	0.678
H2-W2	-0.575	0.991	-4.837	<0.001
H3-W3	3.938	<0.001	-6.337	<0.001

2.3.3 Alexander Range

Possums, mice and rats were recorded during monitoring at Alexander Range (Table 2.7, 2.8). No stoat tracks were recorded. The GLM analysis found a significant season x treatment interaction only for possums, showing different trends at the two sites over time. There was also a significant season effect for possums, and significant site effects for possums, mice and rats (Table 2.9).

Table 2.7. Chew card totals of mammals in the Alexander Range area over three seasons. 'Total' is the number of cards exposed; a card may have signs for several different animals.

Season	**Winter 2012**	**Summer 12/13**	**Summer 13/14**	**Winter 2012**	**Summer 12/13**	**Summer 13/14**
Treatment	Non-tmt	Non-tmt	Non-tmt	Tmt	Tmt	Tmt
Possum	10	13	29	12	0	0
Mouse	0	1	9	8	0	6
Rat	0	0	4	17	4	17
Blank	40	36	10	17	46	28
Total	50	50	50	50	50	50

Table 2.8. Tracking tunnel totals of mammals and weta in the Alexander Range area over three seasons. "Night" is number of nights exposed, and peanut butter pait was used. "Total" is the number of tunnels exposed; a tunnel may have signs for several different animals. "Set" is the exact location of tunnels within the site.

Season	**Winter 2012**	**Summer 12/13**	**Summer 13/14**	**Winter 2012**	**Summer 12/13**	**Summer 13/14**
Treatment	Non-tmt	Non-tmt	Non-tmt	Tmt	Tmt	Tmt
Method	2	1	1	2	1	1
Set	B	A	A	B	A	A
Possum	0	4	1	0	0	0
Mouse	1	0	0	3	0	6
Rat	3	0	3	2	2	16
Cat	0	0	0	0	0	0
Hedgehog	0	0	0	0	0	0
Weta	17	35	34	1	24	14
Blank	18	6	11	13	22	15
Total	39	50	50	19	50	50

Prior to the 1080 operation, possum levels were similar across both sites. Possum levels declined from a 24% detection rate to 0% at the treatment site immediately following the 1080 application (Fig. 2.5, Table 2.10), and no signs of possums were recorded on chew cards or tracking tunnels during either post-1080 season (Table 2.7, 2.8). In contrast, possum levels at the non-treatment site increased from 20% in winter 2012 to 58% in summer 2012/13, to become significantly higher than the treatment site (Fig. 2.5, Table 2.10). There was a significant effect of season following the same pattern as at the Rolleston range (Table 2.9), with possum detection lowest during winter 2012.

Possum detection rate in tracking tunnels was low, with an average rate of 5% over the summer seasons at the non-treatment site, and no detection at the treatment site throughout the study. GLM analysis was therefore not performed for tracking tunnel marks by possums at Alexander Range.

Mice were recorded on chew cards and tracking tunnels at low levels at both sites (Fig. 2.5, Tables 2.7, 2.8). Tracking tunnel data for mice showed a significant site effect, with higher detection rates at the treament site.

Rat signs were recorded from chew cards and tracking tunnels at both sites (Table 2.7, 2.8). Site effects showed that there were significantly higher levels of rats at the treatment site than the non-treatment site across all monitoring periods, with an average detection rate of 35.3% across all three seasons compared to 4.3% at the non-treatment site. There was also an effect of season for chew card detection of rats with levels highest in winter 2012 (34%) before declining to 8% in summer 2012/13, and an average chew card rate of 21% in summer 2013/14 across both sites (Table 2.9).

Table 2.9. Summary of significant predictors for Alexander mammal data in quasipoisson GLMs (poisson GLM for mice in tracking tunnels, P values for the effects listed). For full details of each GLM see Appendix A.

Species	**Season**	**Site**	**Season:treatment**
Mouse, chewcard	0.095	0.590	0.258
Mouse, track tunnel	0.171	**0.013**	0.513
Possum, chewcard	0.176	**<0.001**	**0.004**
Rat, chewcard	**0.038**	**0.006**	0.551
Rat, track tunnel	0.059	**0.048**	0.603

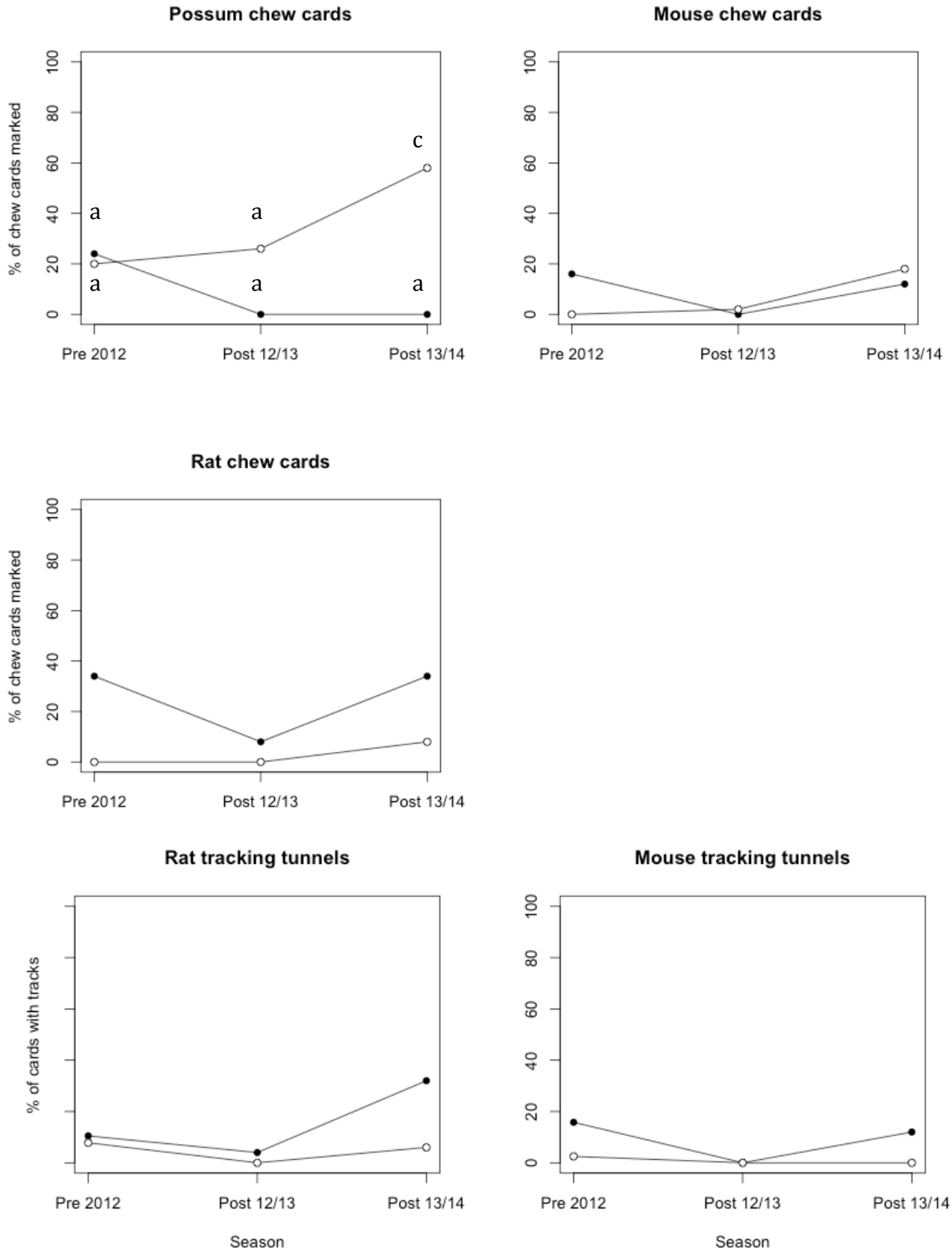

Fig. 2.5. The percentage of marked chew cards and printed tracking tunnels collected across three seasons at Alexander Range for possums, mice and rats. Black circles represent the treatment site, and hollow circles represent the non-treatment site. The letters on the graphs indicate the results of post-hoc tukey analysis, which was performed when a significant

'season x treatment' interaction was present (see tables 2.10a and b); means that are significantly different are represented by different letters. Only possum chew cards required a post-hoc analysis.

Table 2.10a. Z and P values for post-hoc multiple comparisons of means (Tukey contrasts). 'T' or 'A' refers to site/treatment (Taipo or Alexander), and seasons are labelled as 1 = pre-treatment, 2 = post-treatment 2012/13, and 3 = post-treatment 2013/14.
a) Chew card analysis for rats and possums.

	Rat		**Possum**	
Site & season	*Z* value	*P* value	*Z* value	*P* value
T1-T2	0.733	0.974	0.499	0.995
T1-T3	0.733	0.974	2.635	0.076
T2-T3	0.733	0.974	2.227	0.200
A1-A2	-1.762	0.454	-1.808	0.426
A1-A3	0.000	1.000	-1.808	0.426
A2-A3	1.762	0.454	0.000	1.000
T1-A1	-1.762	0.454	-0.338	0.999
T2-A2	0.000	1.000	1.883	0.378
T3-A3	-1.703	0.494	2.816	<0.05

b) Tracking tunnel analysis for rats and mice.

	Rat		**Mouse**	
Site & season	*Z* value	*P* value	*Z* value	*P* value
T1-T2	-0.781	0.968	-0.160	1.000
T1-T3	-0.188	1.000	-0.610	1.000
T2-T3	0.647	0.986	0.000	1.000
A1-A2	-0.665	0.984	-1.864	0.409
A1-A3	1.144	0.853	-0.416	0.998
A2-A3	2.061	0.292	1.727	0.498
T1-A1	-0.258	1.000	-1.669	0.537
T2-A2	-0.383	0.999	0.000	1.000
T3-A3	-1.990	0.331	-1.727	0.498

2.4 Discussion

The aerial 1080 application was successful at reducing possum levels at both treatment sites, with detection rates remaining at or near zero up to 16 months after the 1080 application. Possum levels were significantly lower at both treatment sites relative to the paired non-treatment sites following the 1080 operation, with Rolleston non-treatment detection significantly increasing over both intervals and Alexander non-treatment detection increasing over the second interval. Mice significantly decreased at the Rolleston treatment site, before returning to pre-1080 levels in summer 2013/14. Rats were present at low numbers in both regions, and no treatment effect was detected. Stoats were not observed in either region.

Responses of mammalian predators to the aerial 1080 application

The length of time possum populations take to recover from poisoning operations depends on a number of factors, including the number of individuals surviving the control and the rate of immigration into the area. Immigration of possums from nearby areas can be affected by habitat size and shape, natural boundaries such as rivers, and distance between patches (Cowan 2000). Barlow (1991) modelled the recovery of a possum population after various kill levels from a single control operation, based on a rate of increase (r) of 0.02 and no immigration into the population (Fig. 2.6). After an initial kill rate of 90%, it will take eight years for a possum population to recover to half its pre-control density.

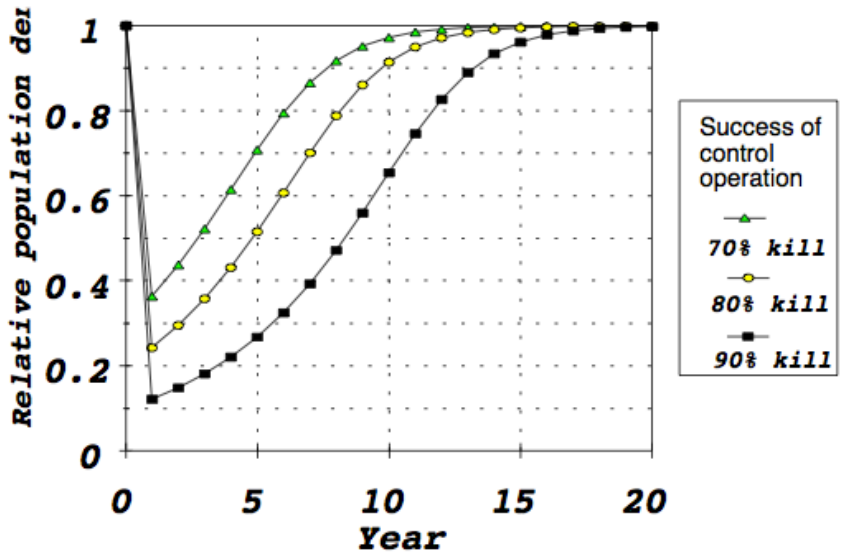

Figure 2.6. Recovery of possum populations with various kill rates, from a single control operation in the absence of immigration (Cowan 2000, based on Barlow 1991 with r=0.2).

Other studies monitoring possum populations after an aerial 1080 application have found that populations with a kill rate over 90% and trap catch indices of near 0 can take between 10 and

20 years to recover to pre-control levels (Coleman et al. 2007, Nugent et al. 2010). TbFree plans to re-treat the Rolleston and Alexander regions with another aerial 1080 application in August 2017, five years after the initial treatment. Given that the detection rate of possums remained at almost zero at both treatment sites during the second year of post-treatment monitoring, it is likely that possum recovery will be slow and that levels will not be able to recover to pre-treatment densities before the next control operation, especially in the Wilberforce area where immigration into the treated forest is restricted by rivers.

As expected for a high altitude mountain beech forest, rat detection was very low at the Rolleston Range sites (King 1983, Kelly et al. 2005). Rats were present at both the treatment and non-treatment sites at Alexander Range, which is to be expected in a lowland mixed native forest habitat (King 2005). Rat detection was low throughout the study period in the paired non-treatment site, and levels fluctuated between 8-34% at the treatment site. Analysis showed no evidence of a significant 1080-related mortality event for rats at the treatment site, although the non-significant change was for a decrease in rats during the first post-1080 counts and a subsequent rebound to pre-treatment levels in summer 2013/14. An increase of rats in response to decreased possum levels as has been occasionally observed in previous monitoring studies (Sweetapple et al. 2006).

Mice were more prevalent at the Rolleston region. The 1080 operation significantly reduced mice levels at the treatment site during the first post-1080 season, but detection rates increased to be higher than pre-1080 levels by summer 2013/14. Mouse detection remained low across all seasons at the non-treatment site, showing no indication of an increase in food availability or other factors that would cause populations to increase independent of the 1080 treatment. As rats and stoats were not detected at either site prior to 1080, the increase in mouse detection cannot be easily explained by release from predation. One possible explanation is that following the eradication of possums, mouse populations were able to quickly recover from 1080-related mortality due to reduced competition with possums for food resources. This has been previously suggested as the reason for an increase in rat numbers following possum removal, as the two species share a dietary overlap for seeds and fruit (Sweetapple & Nugent 2007, Ruscoe et al. 2011). Mice are generalist omnivores, consuming a range of plant material and small invertebrates (King 2005), and may also benefit from a increase in available food supplies following control of possums.

Mammal monitoring methods

Chew cards and tracking tunnels are useful monitoring methods for providing a relative index of mammalian predators, but have several limiting factors that should be taken into account

when drawing conclusions. No stoats were recorded at either region, despite the use of rabbit bait during the pre-treatment monitoring under different nights of exposure. In subsequent seasons, peanut butter bait was used and stoats were not expected to be detected. Gillies and Williams (2013) suggest that tracking tunnels are not always sensitive to mustelids when they are present in very low numbers, and that 'no detection' should be treated with caution. Previous studies have found that mouse tracking rates may not always accurately reflect the density of mice in an area, largely due to behavioural interactions between mice and rats (Sweetapple et al. 2006). In some studies the tracking rate of mice increased after rat eradication, but the abundance of mice measured by trapping was unchanged (Brown et al. 1996, Ruscoe et al. 2011). Throughout this study, rat detection was very low to zero across both regions, so the mouse tracking rates obtained are unlikely to have been affected by altered behaviour due to the presence of high rat densities.

Different methods were used for tracking tunnel monitoring in winter 2012 (two nights exposure) as compared to the two subsequent summer periods (one night exposure), and it is possible this may have some effect on the results obtained. Focussing on the differences between indices of treatment and non-treatment sites over time, rather than the increase or decrease within a single site over time, reduces the chance of method or seasonal factors affecting conclusions. Analysis of different tracking tunnel methods used during winter 2012 suggests that the two methods used should be relatively comparable. Using two monitoring methods also increases the reliability of results, and for most species the patterns obtained from chew cards and tracking tunnels were similar. One exception was a low chew card rate and a much higher tracking tunnel rate for mice at the Rolleston non-treatment site, which could be due to chew card saturation from possums.

The reduction and recovery of mammalian pest populations following an aerial 1080 applicaton in a forest area is crucial to understanding the long term responses of non-target species such as birds and large invertebrates. Monitoring pest levels at the Rolleston and Alexander Ranges can help explain any changes observed in bird and weta populations.

3. Bird monitoring

3.1 Introduction

One of the most common public concerns surrounding aerial 1080 operations is the possibility that individuals of non-target species, such as native birds, will directly or indirectly consume the baits and be poisoned (Green & Rohan 2012). Since 1080 was introduced as a pest control method in 1956, individuals of 19 native and 13 introduced bird species have been found dead after aerial applications of 1080 baits (Spurr 2000, Table 3.1). However, it was not until 1976 that the first formal monitoring of bird deaths attributable to aerial 1080 poisoning was conducted. It was found that birds from a wide range of species were dying in areas where operations were conducted using non-dyed, raspberry-lured carrot baits containing a lot of "chaff", or small fragments (Spurr 2000, Eason et al. 2011). After 1977, several changes were made to the 1080 baits to minimise the risk of poisoning of non-target species. Baits are now dyed green to make them less attractive to birds (Caithness & Williams 1971), carrot baits are screened through a fine grid to remove chaff, and cereal-based baits have become more common as they are thought to kill fewer birds (Spurr 1991). Cinnamon oil is added to both carrot and cereal-based baits to both mask the smell of 1080 from possums, and to deter birds (Spurr & Powlesland 1997).

The effect of 1080 operations on non-target individuals has been well studied for some bird species. Radio tagging or banding individual birds and monitoring their presence before and after aerial 1080 applications in both treatment and non-treatment areas has been used to test for 1080-related mortality in robins, tomtits, kaka, morepork, kokako and kiwi (Spurr & Powlesland 1997, Spurr 2000, Powlesland et al 2000, Veltman & Westbrooke 2011, Greene et al 2013; see Table 3.1 for scientific names). Fewer birds have been reported dead during aerial 1080 applications since the changes to the baits were made after 1977 (Spurr & Powlesland 1997), but the risk of 1080 related mortality is still yet to be quantified at the population level for 11 of the native bird species that are known to have died (Veltman & Westbrooke 2011).

Table 3.1. Bird species found dead after possum control operations using aerial 1080 poisoning (Spurr 2000).

Native species	Scientific name	Introduced species	Scientific name
Harrier	*Circus approximans*	California quail	*Lophortyx californica*
Weka	*Gallirallus australis*	Chukor	*Alectoris chukar*
Pukeko	*Porphyrio porphyrio*	Skylark	*Alauda arvensis*
Black-backed gull	*Larus dominicanus*	Hedge sparrow	*Prunella modularis*
Kereru	*Hemiphaga novaseelandiae*	Blackbird	*Turdus merula*
Kaka	*Nestor meridionalis*	Song thrush	*Turdus philomelos*
Kea	*Nestor notabilis*	Yellowhammer	*Emberiza citrinella*
Morepork	*Ninox novaseelandiae*	Chaffinch	*Fringilla coelebs*
Rifleman	*Acanthisitta chloris*	Greenfinch	*Carduelis chloris*
Pipit	*Anthus novaseelandiae*	Goldfinch	*Carduelis carduelis*
Whitehead	*Mohua albicilla*	Redpoll	*Carduelis flammea*
Grey warbler	*Gerygone igata*	House sparrow	*Passer domesticus*
Fantail	*Rhipidura fuliginosa*	Magpie	*Gymnorhina tibicen*
Tomtit	*Petroica macrocephala*		
Robin	*Petroica australis*		
Silvereye	*Zosterops lateralis*		
Bellbird	*Anthornis melanura*		
Tui	*Prosthemadera novaseelandiae*		
Kokako	*Callaeas cinerea*		

As well as the risk of individual mortality, our understanding of the ability of bird populations to recover from any 1080 related mortality events is important in order to quantify the risk of species being negatively affected. One way to do this is by monitoring the nesting success of non-target species before and after 1080 application. Powlesland et al. (1999) found that an aerial 1080 application in 1996 using carrot-based bait with a large amount of chaff resulted in a 55% mortality rate of banded North Island robins. Despite this, the reduction in possum and rat densities in treatment areas during the nesting season meant that robin breeding success was much higher than in the non-treatment areas. A year after treatment, population numbers of robins in the treatment area had showed a 125% increase, including mortality from 1080, from numbers recorded a month after the 1080 operation. In contrast, species that have lower reproductive rates may be less able to recover from a 1080-related mortality event. Nevertheless, vulnerable species such as kokako, North Island kiwi, great spotted kiwi, and blue duck have been monitored throughout aerial 1080 operations, and there was no indication of mortality being high enough to threaten the viability of a population (Eason et al. 2011).

Although the monitoring of marked individuals (such as radio-tagged or colour-banded birds) is important to determine the short-term mortality rates of aerial 1080 applications in non-target species, long-term monitoring is necessary for understanding the net positive or negative effects that 1080 may have on non-target species over months or years. Veltman & Westbrooke (2011) advocated for the establishment of long-term monitoring of forest bird populations at sites treated with 1080 in order to allow measurement of rates of changes in population sizes, and thus provide more information to managers about when and how often the operations should occur.

The most comprehensive study on the long-term benefits of pest control for forest bird populations in New Zealand was conducted by O'Donnell & Hoare (2012). They analysed information from 12 years of bird and pest monitoring in the Landsborough Valley, where continual trapping was used to control mustelids and pulsed aerial 1080 applications (around every 2-5 years) were used for possum and rat control. Annual five-minute bird counts showed that nine species (bellbird, brown creeper, fantail, grey warbler, mohua, rifleman, tui, yellow-crowned parakeet and song thrush) showed significant increases during the study period. In contrast, four species (silvereye, tomtit, chaffinch and blackbird) significantly declined during the study, although there was no matched non-treatment area to rule out other factors contributing to these patterns. In any case, of the decreasing species, two are exotics and one (silvereyes) is very widespread and abundant, so local toxin impacts on those three would be of less concern. O'Donnell & Hoare (2012) concluded that multiple species benefit from integrated pest control, but the pattern of response can vary between different species.

Predicting species responses

Factors contributing to the long-term responses of bird species to 1080 applications could include variation in 1080 poisoning risk, predation vulnerability, and the reproductive rates of the species, as well as other site and species-specific factors. Table 3.2 summarises ecological trait and 1080 poisoning information for 12 common native and introduced forest bird species. During the first breeding season following a successful aerial-1080 application, bird species that have high reproductive rates but are limited by mammalian predation would be expected to show a population increase. Frugivoruous, insectivorous, herbivorous and nectivorous birds such as kokako, kereru, tui and bellbird could also benefit from increased food resources when possum numbers are low (Innes et al 2010).

Blackbird, kereru, robin and tomtit individuals have all been observed feeding on 1080-laced cereal baits, so may be at a higher risk than other bird species of suffering from a 1080-related mortality event (Powlesland et al. 1999, Powlesland et al. 2000, Spurr 2000). However, robin

and tomtit populations have been found previously to recover quickly from 1080-related mortality events due to elevated breeding successes in the absence of mammalian predators (Powlesland et al. 1999, 2000). Therefore, any 1080-related mortality events that do occur in these species could be compensated for by an increased reproductive success during the following breeding season.

Objectives

This chapter presents the results of bird monitoring. The purpose of the monitoring was to determine how bird species responded to a one-hit control of aerial 1080, using five-minute bird counts to measure any changes in relative bird abundance. The monitoring was conducted over three seasons (one pre-treatment and two post-treatment) at the Rolleston and Alexander Ranges. Relative abundances of bird populations were compared between treatment and paired non-treatment sites and over time.

Predictions of the response of bird populations to the 1080 operation were as follows:

1) If any species suffers from direct 1080-related mortality, this would be seen as a decrease in the first post-1080 monitoring season in the treatment site relative to the non-treatment site.
2) Bird species which normally suffer major impacts from introduced predators should benefit from the decrease in mammalian predators, and increases should be most clearly detectable in bird species with high reproductive rates.
3) Birds most sensitive to possums should increase over both post-1080 seasons, whereas birds sensitive to rodent predation should increase over the first season but may decline in the second season if rodents rebound above initial density.

Table 3.2. Trait information and 1080 poisoning information for 12 common forest bird species (modified from Hoare et al. 2012). Foraging, fecundity and predation vulnerability is based on information from Heather & Robertson (2005). Individual monitoring before/after an aerial 1080 application refers to whether the species has been studied using radio or band tagging methods (Spurr & Powlesland 1997, Armstrong & Ewen 2000, Powlesland et al 2000). Change in abundance refers to a study by Elliot et al. (2010), where bird count information spanning 30 years from Lake Rotoroa, in Nelson Lakes National Park was analysed to determine whether common birds were declining in undisturbed Nothofagus forest with no management of pests (rodents, stoats and possums).

Species	**Poisoning risk**		**Foraging**		**Fecundity**			**Predation vulnerability**		**Change in abundance**
	Known to eat 1080 cereal baits	Individual monitoring b/a 1080	Diet	Foraging strata	No. of eggs	Max. clutches	Age at first breeding	Nest location	Length of nesting cycle (days)	Over 30 years (Elliot et al. 2010)
Bellbird	N	N	Omnivore	Throughout	4	2	1	Cavity & open	28	Decline
Blackbird	Y	N	Omnivore	Ground	4	5	1	Open	29	No change
Brown creeper	N	N	Insectivore	Canopy	3	2	1	Open	43	No change
Chaffinch	Unknown	N	Granivore							-
Fantail	N*	N	Insectivore	Throughout	4	5	1	Open	32	No change
Grey warbler	N*	N	Insectivore	Throughout	4	3	1	Open	40	Decline
Kereru	Y	N	Herbivore	Canopy	1	3	1.5	Open	58	-
Rifleman	N*	N	Insectivore	Throughout	4	2	1	Cavity	47	Decline
Robin	Y	Y	Insectivore	Ground	3	?	1	Open	39	Increase
Silvereye	N*	N	Omnivore	Throughout	3	3	1	Open	23	Increase
Tomtit	Y	Y	Insectivore	Canopy, ground	3	3	1	Cavity	37	Decline
Tui	N*	N	Onmivore	Throughout	3	?	?	Open	35	Decline

* have been known to eat 1080 carrot-based baits.

3.2 Methods

3.2.1 Five-minute bird counts

Bird monitoring was conducted using the five-minute bird count method, as described by Dawson & Bull (1975). This method has been consistently used in New Zealand for almost 40 years, and allows consistent quantitative information to be collected for multiple species of birds (Hartley 2012). The method consists of an observer standing stationary at a specific point (in this case a permanently marked station), and counting every bird of every species that they see or hear within a five minute time period. Weather conditions such as sun, wind, rain and noise (such as nearby streams) are also recorded for each count. For our study a maximum radius of 100m was used, beyond which birds were not included in the count.

Permanent bird count stations were established at each site (see Chapter 2, Fig. 2.1 and 2.2 for maps). Count stations were interspersed with tracking tunnels, using every second station for a 200m bird count spacing. Each site had 20 bird count stations. The total number of counts conducted at each site per season ranged from 40 to 179 (table 3.4), with the target being at least 80 counts. Only 40 counts were conducted at the Alexander site in winter 2012 due to weather conditions.

All five minute bird counts were conducted by teams from the University of Canterbury: Josh Van Vianen, Alwin Sky and James Mortimer in winter 2012, Josh Van Vianen, Archie Macfarlane and myself in summer 2012/13, and Josh Van Vianen, Archie Macfarlane and occasionally myself in summer 2013/14.

Table 3.3. Field work dates for five-minute bird counts.

	Rolleston Range		**Alexander Range**	
	Wilberforce	Harper	Alexander	Taipo
Winter 2012	28-30 June	20-22 July	5-6 July	26-28 July
Summer 2012/13	29-31 Jan	21-24 Jan	18-20 Dec	11-14 Dec
Summer 2013/14	21-24 Jan	14-17 Jan	3-6 Dec	3-6 Dec

3.2.2 Statistical analysis

Each bird species was analysed separately, and the Rolleston and Alexander regions were analysed separately. The analysis for bird counts for each species used a Generalised Linear Model in R (version 2.10), with a Poisson error distribution and fixed terms for season, site, and a season x treatment interaction. The nested nature of the data (repeat counts over several days at each station, repeat stations within each site) was accounted for by totaling counts at the station level (i.e. adding together the total number of individuals of each species recorded

over the 3 or 4 counts at each station per season) and using the command '*offset*' in R with the number of counts per station to gain an average count number for each species per station. A significant season x treatment interaction indicates that the treatment site has changed over time in a different way to the non-treatment site, showing an effect of the aerial 1080 operation. If this interaction was significant, another GLM was run with the fixed term 'sitedate', which had separate values for each treatment and season combination (eg. Rolleston winter 2012, Alexander summer 2013 etc). This was then used for post-hoc means testing with a multiple means comparison (Tukey contrasts, using library multcomp and the command *glht*) to determine where over sites and seasons the changes occurred.

As with the mammal data, the problem with a large number of zeros causing a "complete separation" error was encountered for some of the rarer species which otherwise had large enough numbers to analyse with the GLMs. This was dealt with the same way as for the mammal analysis (i.e. a '1' count was added to groups which contained all zeros, so the analysis is conservative).

Some corrections were made to the data to allow for observer discrepancies. One observer during the winter 2012 season at the Wilberforce area, James Mortimer, had impaired high-frequency hearing which reduced his detection rate of rifleman calls. The analysis for rifleman at Rolleston Range included a term for observer (James vs. other) and presents the fitted values expected for "other" observers. This meant that the rifleman analysis could not use totals of each station during a season as above, as the observer was not the same for all of each station's counts. Instead the analysis was carried out at an individual count level, using a Poisson GLMM (library lme4, command 'lmer'), with a random term for count station and fixed effects for season, site, treatment and observer.

During summer 2013/14 monitoring, after observing a number of cuckoos, the observers (Josh Van Vianen and Archie Macfarlane) realized that they had been recording long-tailed cuckoo calls as greenfinches, which have similar calls. Prior counts recorded as "greenfinch" have therefore been changed to long-tailed cuckoo counts.

3.2.3 Power analysis

The power of a statistical test is the probability that the test will correctly reject a false null hypothesis (Steidl et al. 1997). Statistical power is a function of the size of the effect, i.e. in this case the difference in bird abundance between sites or over time, and the sample size. Power analyses can be used *a priori* to estimate the number of samples needed to achieve a high probability of detecting a biologically significant effect, or can be used after data

collection to estimate the detectable effect size that can be obtained from the data. Most researchers consider a power of 0.8 or higher to be acceptable (Choquenot & Ruscoe 1999).

The statistical power of the five-minute bird count technique was first estimated by Dawson (1981). He proposed that unless a bird species is very abundant (an average of >1 birds per count) and large numbers of counts are made (>30 in each site), the technique has low power to detect changes of less than 50% in forest bird populations. It is possible to calculate the power for detecting a difference between means of count point data from two treatments using Dawson's equation:

$$number\ of\ stations > \frac{(b\ x\ 20000)}{(d^2\ x\ m)}$$

where b is a value that corresponds to the desired power, and d is the actual or desired effect size, defined as:

$$d = 100\left[\frac{(m_1 - m_2)}{m}\right]$$

where m1 and m2 are the means in each area, and m is the overall mean for both areas. The formula assumes a Poisson distribution of the counts.

I used this formula to estimate the number of bird counts needed to detect effect sizes ranging from a 20-90% change in populations for bird species with varying average abundances (Table 3.4). The power level was specified at 0.8.

Power for Rolleston bird analysis

Dawson (1981) states that if a different number of counts are made in each sample, the average count number of the samples can be used providing that the smallest sample is at least half the size of the largest one. The lowest average number of counts in the Rolleston area was 95 during winter 2012 at the treatment site, and the overall average number of counts across all monitoring periods was 117 (Table 3.5). Therefore, comparisons between winter samples will likely have high power to detect a 50% change in species with up to 0.8 birds per count and a 30% change in very abundant species (2 birds per count), but will have lower power for less common birds. For species as rare as 0.2 per count, even a 90% change will have low power due to the small sample size. Changes as low as 30% may be able to be detected with high power for abundant species (>1.5 per count) within the summer seasons.

Table 3.4. The number of bird counts needed in each area to achieve power of 0.8 for various levels of effect size and bird abundance. Shaded areas indicate the number of counts that have been achieved for this study in one or both regions.

	Effect size (%)							
No. birds per count	**20**	**30**	**40**	**50**	**60**	**70**	**80**	**90**
0.2	>500	>500	500	320	223	164	125	99
0.4	>500	445	250	160	112	82	63	50
0.6	>500	297	167	107	74	55	42	33
0.8	500	222	125	80	56	41	31	25
1.0	400	178	100	64	45	33	25	20
1.20	334	148	84	54	37	27	21	17
1.40	286	127	72	46	32	23	18	14
1.60	251	111	63	40	28	20	16	12
1.80	223	99	56	36	25	18	14	11
2.00	200	89	50	32	22	16	13	10

Table 3.5. The total and average numbers of five-minute bird counts achieved in each region over three seasons.

	Rolleston Range			**Alexander Range**		
	Wilberforce	Harper	Average	Alexander	Taipo	Average
Winter 2012	80	110	95	40	75	57
Summer 2012/13	100	110	105	110	98	104
Summer 2013/14	130	174	152	120	179	149
Total average			117			103

Power for Alexander bird analysis

As an average of only 57 counts were achieved during winter 2012 in the treatment site at Alexander, comparisons within this season will have only have the power to detect changes of 50% or above, especially for less abundant bird species. However, comparisons within the summer seasons may be able to detect changes of 30% for abundant species and at least 50% for species as rare as 0.6 birds per count.

3.3 Results

3.3.1 Rolleston Range

A total of 24 bird species were recorded during the monitoring at Rolleston Range; 13 native species, and 11 introduced species. Eight of the most abundant species (based on means, Table 3.6) were chosen for analysis with GLMs: bellbird, brown creeper, silvereye, rifleman, tomtit, grey warbler, chaffinch and blackbird. Counts for the other species were too low to allow statistical analysis, but their presence is still of interest.

Table 3.6. The average number of birds recorded per five-minute count at Rolleston Range across three seasons at the treatment and non-treatment sites.

Species	Treatment			Non-treatment		
	Winter 12	Summer 12/13	Summer 13/14	Winter 12	Summer 12/13	Summer 13/14
Bellbird	1.75	1.33	2.30	0.75	0.77	0.90
Brown creeper	0.1	0.60	0.72	1.15	2.26	2.20
Silvereye	0.70	1.74	2.04	0.68	1.43	1.94
Chaffinch	0.15	2.04	0.98	0.55	2.17	1.91
Rifleman	1.30	1.28	1.41	1.59	1.77	1.84
Tomtit	0.04	1.19	0.98	0.09	0.26	0.12
Redpoll	0	0.05	1.02	0	0	0.8
Grey warbler	0	0.54	0.58	0.04	0.54	0.83
Blackbird	0.16	0.64	0.74	0.02	0.21	0.27
Song thrush	0	0.22	0.29	0	0.02	0.01
Long-tailed cuckoo	0	0.24	0.22	0	0.34	0.17
Fantail	0	0.20	0.06	0.02	0.1	0.07
Greenfinch	0	0	0.03	0	0	0.01
Finch	0	0.12	0.02	0	0.19	0.05
Falcon	0	0.01	0.02	0	0	0.02
Kea	0	0.12	0	0.01	0.11	0.09
Dunnock	0.13	0	0.02	0	0.12	0.09
Goldfinch	0	0.01	0.03	0	0.08	0
Harrier	0	0	0	0	0	0
Kereru	0	0	0.01	0	0	0
Black-backed gull	0	0	0.02	0	0	0
Swallow	0	0	0	0	0	0.04
Magpie	0	0	0	0	0	0.01
Tui	0	0.01	0	0	0.04	0.01

"Season" effects (where changes occurred over time at both sites, Table 3.7) showed that bird counts were generally lower in the winter 2012 period than during the summer periods. This is likely due to seasonal changes in bird visibility, and doesn't necessarily reflect changes in bird numbers. Some bird species are less conspicuous during winter, and become more detectable by observers in summer during the breeding season and with the addition of juveniles in the population (Dawson 1978). Migratory behaviour can also explain seasonal variability for some species such as the long-tailed cuckoo, which were not detected during winter 2012 as they are only present in New Zealand forests between October and February (Heather & Robertson 2005).

Site effects were significant when a constant difference in counts of a species was observed between the matched sites, and varied among bird species (Table 3.7). Counts of bellbirds and blackbirds were higher at the treatment site, whereas brown creeper and chaffinch levels were higher at the non-treatment site.

A significant "season x treatment" interaction indicates a relative shift in bird levels between the treatment and non-treatment sites over time. Overall, the analyses showed that two species (tomtit and chaffinch) had a significant season x treatment interaction, with different trends over time observed in the two sites (Table 3.7). Tomtits increased in response to the 1080 treatment, whereas chaffinches decreased between post-1080 seasons (Fig. 3.1).

Table 3.7. Analysis of five-minute bird counts at the Rolleston study site: significance tests from quasi-Poisson GLMs, and Poisson GLMM for rifleman (P value for the effect listed). For full details of each GLM see Appendix B. NS = not significant.

Species	Season	Site	Season:treatment
Bellbird	0.015	<0.001	0.405
Brown creeper	0.002	<0.001	0.266
Silvereye	<0.001	0.334	0.877
Chaffinch	<0.001	<0.001	**0.012**
Grey warbler*	<0.001	0.037	0.270
Blackbird*	<0.001	<0.001	0.229
Tomtit*	<0.001	<0.001	**<0.001**
Rifleman	NS	NS	NS

*Poisson (no overdispersion)

Tomtits at the treatment site responded to the 1080 application with a significantly larger increase than in the non-treatment site, with counts increasing 29.8 fold at the treatment site over the summer seasons compared to 2.9 fold at the non-treatment site. (Fig. 3.1a). Seasonal variation in detectability was observed, with tomtits at both sites recorded at low counts over winter 2012. During summer 2012/13 counts at both sites had increased, suggesting that tomtits were less visible to observers during winter. However, levels at the treatment site increased significantly more than at the non-treatment site, with an average of 1.19 tomtits per count compared to 0.26 per count at the non-treatment site (Table 3.6). Counts in summer 2013/14 remained high at the treatment site, and lowered slightly at the non-treatment site.

Chaffinches also had a significant season x treatment interaction, but showed a different pattern of response (Fig. 3.1b). Chaffinches at both sites were recorded at low levels prior to the 1080 application. By summer 2012/13 the counts at both sites were equally higher, suggesting that chaffinches were also less visible during winter. However by the second post-1080 season, chaffinches at the treatment site had decreased by more than half but did not do

so at the non-treatment site, making the difference between the two sites significant (Table 3.7).

Silvereyes had the highest average count across all seasons, meaning that a size of change as low as 27% could be detected with high power between samples (Table 3.8). Changes between 30-34% could be detected for the other relatively abundant species (rifleman, chaffinch, bellbird, brown creeper). Tomtits, grey warblers and blackbirds had the lowest means, and only a change of at least 63% could be detected with high power for blackbirds (Table 3.8).

Table 3.8. The mean abundance of the eight most common bird species across all three seasons at Rolleston Range, and the minimum size of change that can be detected with 80% power (mean station counts across all three seasons = 117). Also listed is whether the species showed a significant 'season x treatment' interaction.

Species	Mean abundance	Detectable size of change	Significant interaction
Silvereye	1.83	27%	No
Rifleman	1.53	30%	No
Chaffinch	1.30	32%	Yes
Bellbird	1.30	32%	No
Brown creeper	1.16	34%	No
Tomtit	0.45	55%	Yes
Grey warbler	0.42	57%	No
Blackbird	0.34	63%	No

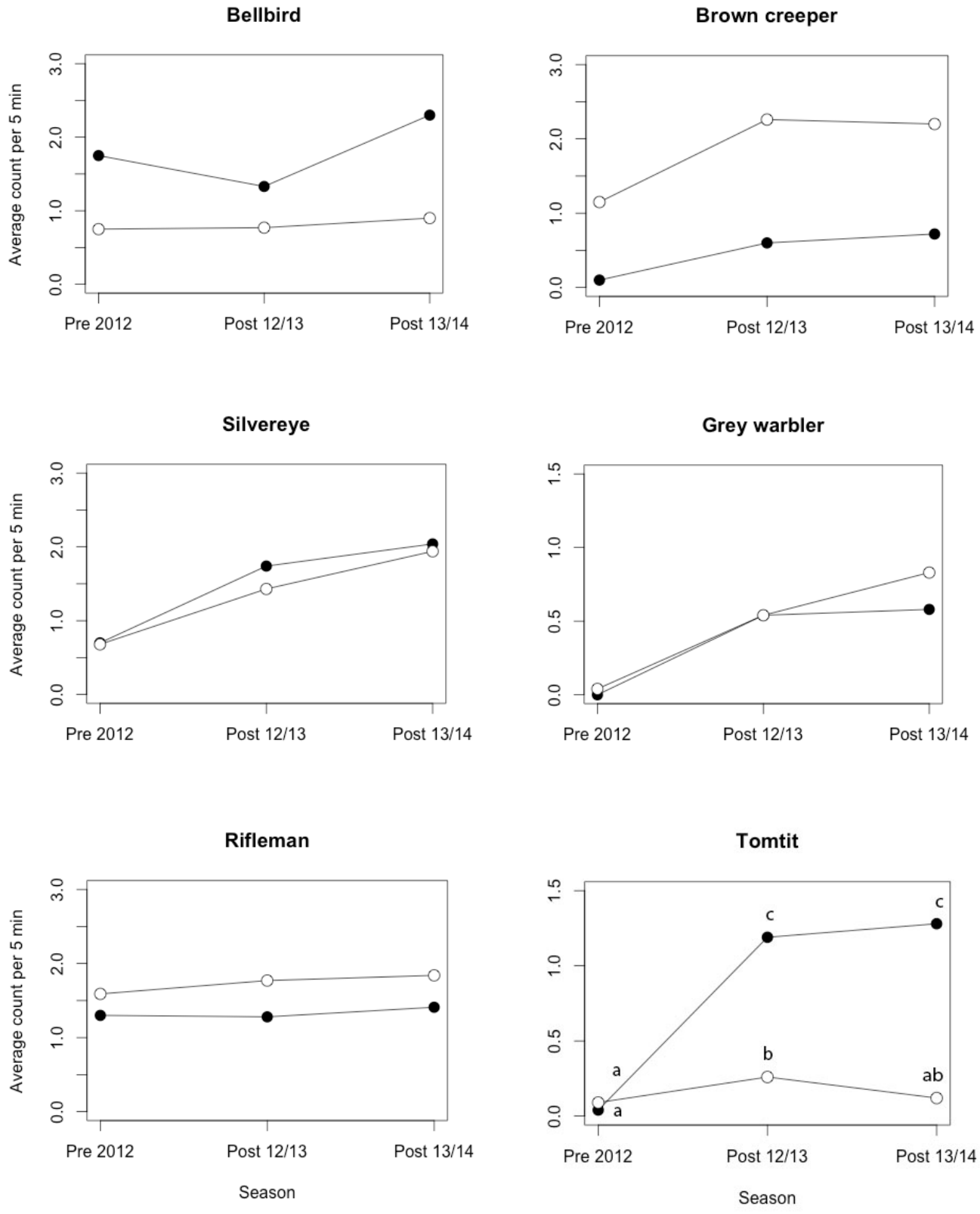

Figure. 3.1. a) Average count of native birds per five-minute count for the most common species at Rolleston Range in the treatment area (black circles) and non-treatment area (hollow circles). The letters on the graphs indicate the results of post-hoc Tukey means comparisons for the two species with significant interactions (Table 3.9); means that are significantly different are represented by different letters.

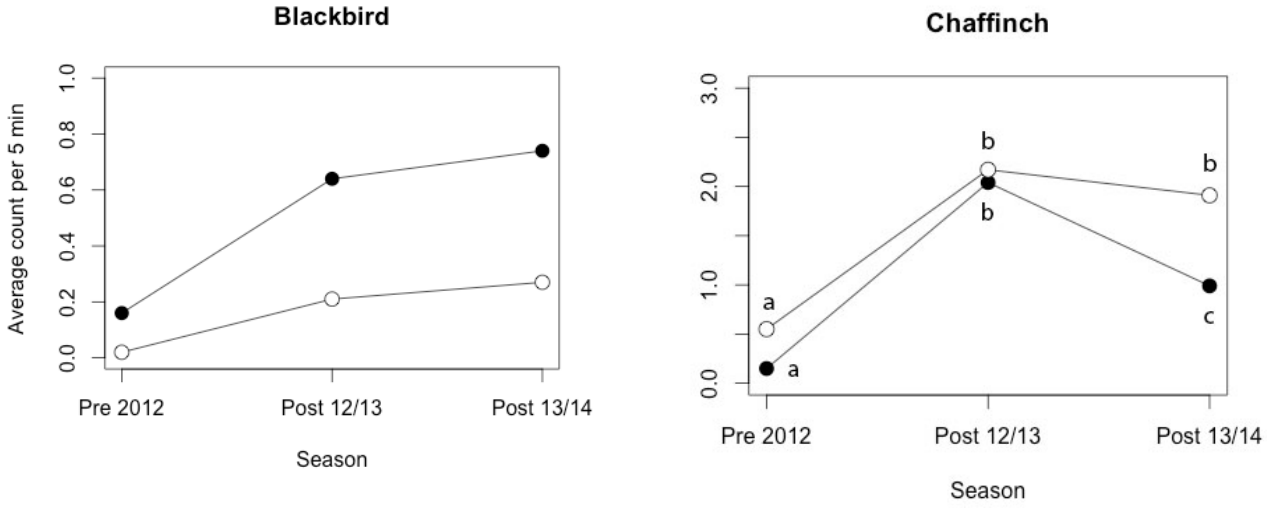

Figure. 3.1. b) Average count of introduced birds per five-minute count for the most common species at Rolleston Range in the treatment area (black circles) and non-treatment area (hollow circles). The letters on the graphs indicate the results of post-hoc Tukey means comparisons for the two species with significant interactions (Table 3.9); means that are significantly different are represented by different letters.

Table 3.9. Z and P values for post-hoc multiple comparisons of means (Tukey contrasts) for species with a significant interaction. 'H' or 'W' refers to site/treatment (Harper or Wilberforce), and seasons are labelled as 1 = pre-treatment, 2 = post-treatment 2012/13, and 3 = post-treatment 2013/14.

	Chaffinch		**Tomtit**	
Site & season	*Z* value	*P* value	*Z* value	*P* value
H1-H2	5.777	<0.001	2.903	0.035
H1-H3	5.405	<0.001	0.783	0.964
H2-H3	-0.884	0.943	-2.666	0.06
W1-W2	5.224	<0.001	5.893	<0.001
W1-W3	3.784	<0.001	5.600	<0.001
W2-W3	-3.633	<0.001	-1.366	0.7122
H1-W1	-2.440	0.125	-1.326	0.7373
H2-W2	-0.552	0.993	7.278	<0.001
H3-W3	-3.737	0.002	8.907	<0.001

3.3.2 Alexander Range

A total of 25 bird species were recorded during monitoring at Alexander Range, with 17 native and eight introduced species. The eight most common species (Table 3.10) were chosen for analysis with GLMs; bellbird, brown creeper, silvereye, chaffinch, grey warbler, tui, tomtit and rifleman.

Tomtit counts reached higher numbers at Alexander, with an average of up to 2.68 tomtits per count (Table 3.10) compared to 1.19 at Rolleston (Table 3.6). Bellbird and grey warbler averages were slightly higher overall at the Alexander region. Rifleman counts were lower, with an average of up to 0.90 counts at Alexander and 1.84 at Rolleston. Brown creepers

were also less common (up to 0.85 per count at Alexander compared to 2.26 at Rolleston), which could account for the lower detection of their nest parasites, the long-tailed cuckoos. Tui were common at Alexander, whereas none were detected at Rolleston.

Table 3.10. The average number of birds recorded per five-minute count at Alexander Range across three seasons at the treatment and non-treatment sites.

Species	**Treatment**			**Non-treatment**		
	Winter 12	Summer 12/13	Summer 13/14	Winter 12	Summer 12/13	Summer 13/14
Tomtit	0.18	0.81	1.63	0.72	2.68	2.66
Bellbird	1.25	0.62	2.61	1.78	0.91	1.63
Grey warbler	0.50	1.04	0.84	0.76	0.94	0.84
Rifleman	0.73	0.53	0.48	0.90	0.33	0.27
Chaffinch	1.48	0.39	0.89	0.43	0.03	0.09
Silvereye	0.08	0.59	1.26	0.68	0.78	1.18
Brown creeper	0.85	0.63	0.26	0.28	0.18	0.28
Tui	0	1.09	0.74	0.01	0.49	0.43
Blackbird	0.08	0.69	1.31	0	0.89	1.19
Fantail	0.08	0.43	0.31	0	0.1	0.12
Kereru	0.28	0.12	0.21	0.01	0	0
Kea	0	0.02	0.05	0.10	0.06	0.05
Song thrush	0	0.01	0.21	0	0.01	0.04
Weka	0.05	0.01	0.13	0.01	0.01	0
Finch	0	0	0.06	0	0.06	0
Greenfinch	0	0.05	0.05	0	0	0
Falcon	0	0	0	0.01	0.02	0
Robin	0	0.01	0.01	0	0	0
Long-tailed cuckoo	0	0	0.01	0	0	0.02
Shining cuckoo	0	0.03	0.03	0	0	0
Yellow-crowned parakeet	0.05	0.04	0.01	0	0.04	0
Hedge sparrow	0	0	0.01	0	0	0
Skylark	0	0	0.01	0	0	0
Paradise duck	0	0	0.01	0	0	0
Starling	0	0	0.01	0	0	0

The GLM analyses (Table 3.11) showed that four species had a significant season x treatment interaction: bellbird, silvereye, tomtit and chaffinch. The season effect was significant for six species, while site was significant for four. For the species with a significant season effect, silvereyes, tui and tomtits had higher average counts during both summer seasons than during winter 2012 (Fig. 3.2). Rifleman, bellbird and chaffinch counts were higher in winter than summer 2012/13. Four species showed a significant site effect, with tomtits more common at the non-treatment site and brown creepers, chaffinches and tui more common at the treatment site.

Fantail and kereru were not common enough to analyse with GLMs. Both species were observed more at the treatment site, and were rarely observed at the non-treatment site (Table 3.10). All other species were observed at very low levels.

Table 3.11. Alexander Range five-minute bird count analysis: significance tests from quasi-Poisson GLMs, and Poisson GLMs for tui and tomtit (P value for the effect listed). For full details of each GLM see Appendix B.

Species	Season	Site	Season:treatment
Bellbird	<0.001	0.251	**0.009**
Brown creeper	0.203	0.008	0.113
Silvereye	<0.001	0.437	**0.023**
Chaffinch	<0.001	<0.001	**0.012**
Grey warbler	0.072	0.837	0.274
Tui	<0.001	<0.001	0.078
Tomtit	<0.001	<0.001	**<0.001**
Rifleman	0.002	0.145	0.112

The species with significant season x treatment interactions showed three different patterns. Bellbirds and silvereyes had similar counts recorded at both sites prior to 1080 and during the first post-1080 season, but counts increased significantly more at the treatment site in the second post-1080 season (Table 3.10, Fig. 3.2). For both bellbirds and silvereyes the average count was higher at the non-treatment site during winter 2012, but higher at the treatment site during the second post-1080 season (Table 3.10, Fig. 3.2). Silvereye counts increased 15.7 fold over the monitoring period at the treatment site, compared to 1.73 fold at the non-treatment site. Bellbird counts doubled at the treatment site over the monitoring period, and decreased slightly at the non-treatment site.

Tomtit counts increased significantly from winter 2012 to summer 2012/13 at both sites, with a 4.5 fold increase at the treatment site and a 3.7 fold increase at the non-treatment site. Only tomtit counts at the treatment site continued to increase in summer 2013/14 (Table 3.10, Fig. 3.2).

Chaffinch counts decreased at both sites from winter 2012 to summer 2012/13, with a 3.8 fold decrease at the treatment site and a 14.3 fold decrease at the non-treatment site. Chaffinch counts then increased 2.3 fold only at the treatment site in the second post-1080 season (Table 3.10, Fig. 3.2).

The sizes of changes able to be detected with high power ranged from 33% for the species with the highest average counts (tomtits and bellbirds), to 54-61% for the species with the lowest average counts (rifleman, tui and brown creeper, Table 3.12).

Table 3.12. The mean abundance of the eight most common bird species across all three seasons at Alexander Range, and the minimum size of change that can be detected with 80% power (mean station counts across all three seasons = 103). Also listed is whether the species showed a significant 'season x treatment' interaction.

Species	Mean abundance	Detectable size of change	Significant interaction
Tomtit	1.45	33%	Yes
Bellbird	1.45	33%	Yes
Grey warbler	0.82	44%	No
Silvereye	0.76	45%	Yes
Chaffinch	0.55	53%	Yes
Rifleman	0.54	54%	No
Tui	0.46	58%	No
Brown creeper	0.41	61%	No

Table 3.13. Z and P values for post-hoc multiple comparisons of means (Tukey contrasts) for species with a significant interaction. 'T' or 'A' refers to site/treatment (Taipo or Alexander), and seasons are labelled as 1 = pre-treatment, 2 = post-treatment 2012/13, and 3 = post-treatment 2013/14.

	Tomtit		**Bellbird**		**Chaffinch**		**Silvereye**	
Site & season	*Z* value	*P* value	*Z* value	*P* value	*Z* value	*P* value	*Z* value	*P* value
T1-T2	8.189	<0.001	-2.704	0.06	-2.974	<0.05	0.416	0.998
T1-T3	8.797	<0.001	-0.502	0.995	-3.015	<0.05	1.711	0.484
T2-T3	0.730	0.973	0.234	0.130	1.139	0.845	1.444	0.666
A1-A2	3.917	<0.001	-2.086	0.279	-3.992	<0.001	1.787	0.434
A1-A3	5.806	<0.001	2.501	0.115	-1.956	0.331	2.545	0.09
A2-A3	5.583	<0.001	5.779	<0.001	2.894	0.036	2.938	0.032
T1-A1	3.521	0.004	1.178	0.837	-3.449	0.006	1.927	0.347
T2-A2	8.907	<0.001	1.333	0.753	-2.928	0.033	0.955	0.920
T3-A3	5.899	<0.001	-2.715	0.06	-4.826	<0.001	-0.511	0.994

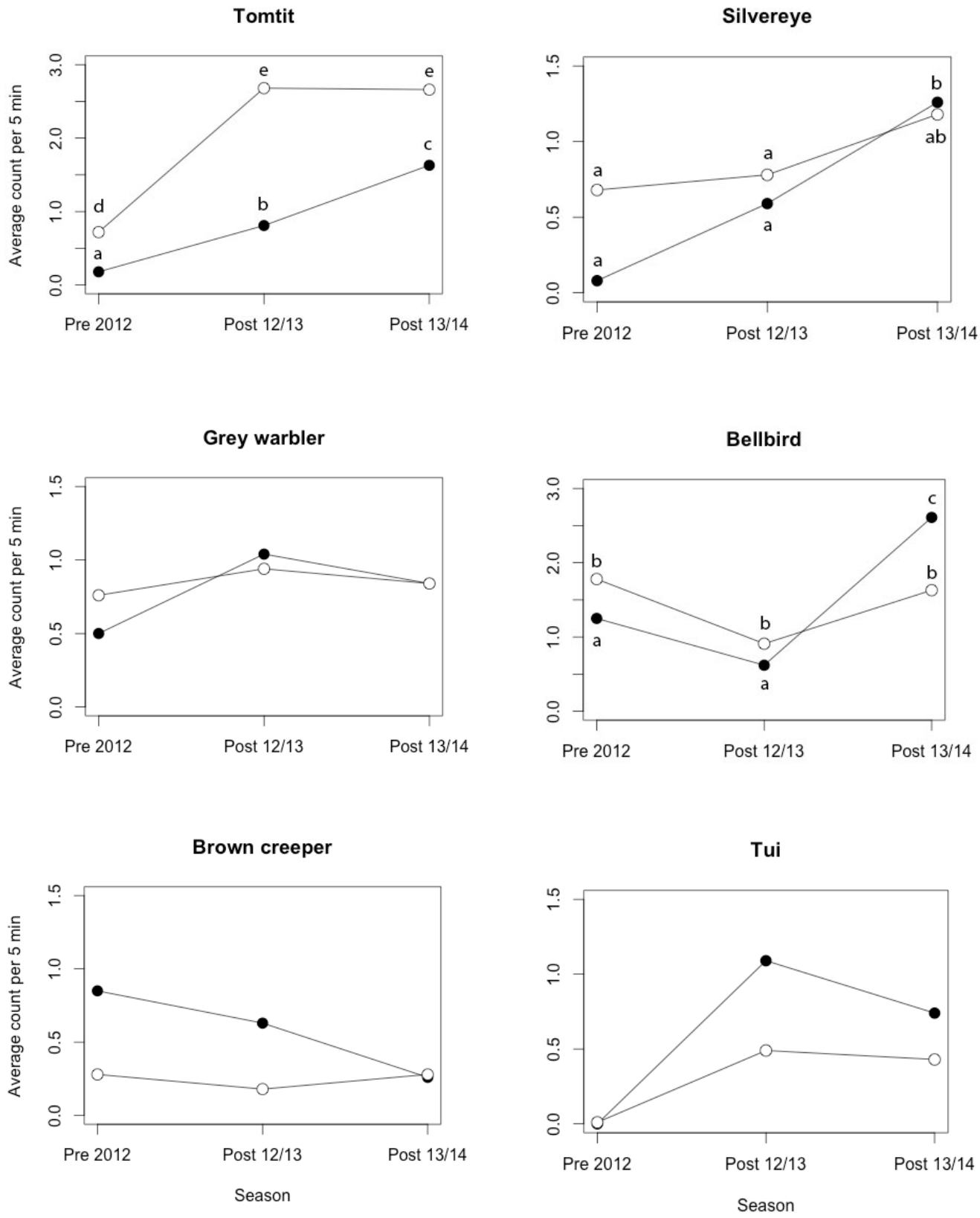

Fig. 3.2. Average number of birds per five-minute count for the most common species at Alexander Range in the treatment area (black circles) and non-treatment area (hollow circles). The letters on the graphs indicate the results of post-hoc Tukey means comparisons for the four species with a significant interaction (Table 3.13); means that are significantly different are represented by different letters.

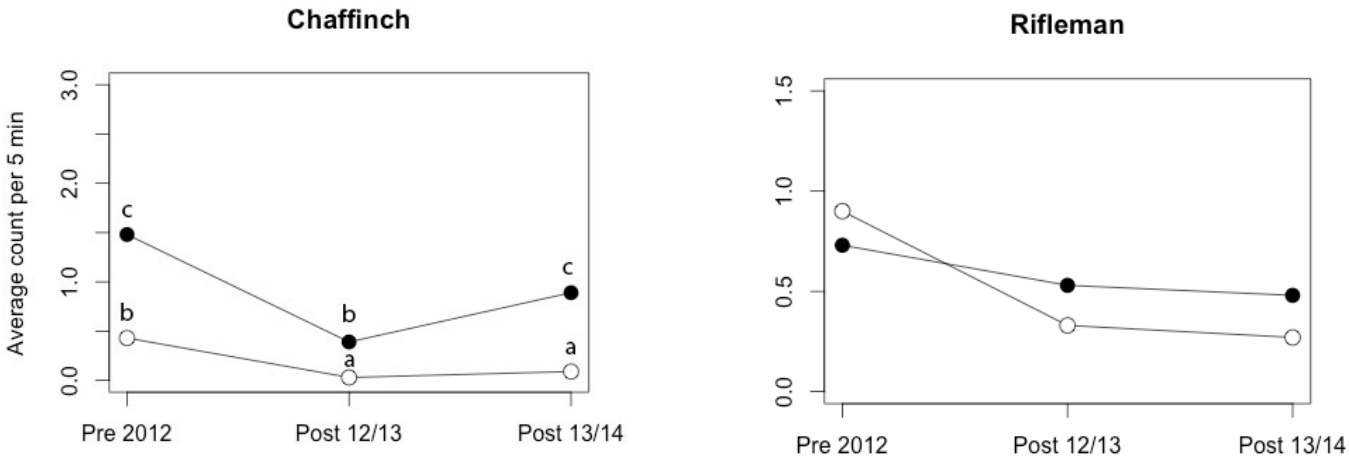

Fig. 3.2 (continued). Average number of birds per five-minute count for the most common species at Alexander Range in the treatment area (black circles) and non-treatment area (hollow circles). The letters on the graphs indicate the results of post-hoc Tukey means comparisons for the four species with a significant interaction (Table 3.13); means that are significantly different are represented by different letters.

3.4 Discussion

Of the nine most common bird species chosen for analysis, all except brown creepers have had dead individuals reported after an aerial 1080 operation (Spurr 2000). It was not possible during this study to monitor the mortality of individuals, but five-minute bird counts are able to estimate the relative abundance of species (Dawson & Bull 1981, Hartley 2012). If a significant 1080-related mortality event occurred for a species, this would be seen as a decrease in relative abundance in the treatment site, but not the non-treatment site, following the 1080 operation. The results of bird monitoring at the Rolleston and Alexander Ranges showed that no native species showed a significant decline in relative abundance immediately following the 1080 operation which was not also seen at the non-treatment sites, or that couldn't be explained by a season or site effect. Tomtits showed an increase in response to the 1080 operation, with counts at both treatment sites significantly increasing in comparison with the non-treatment sites.

Seasonal changes in conspicuousness

Brown creeper, silvereye, grey warbler, blackbird, tomtit and tui counts were all lower during winter 2012 at one or both regions. These seasonal changes in conspicuousness are commonly observed. Dawson (1978) found that for most common forest bird species, numbers counted were lowest during winter and highest during summer due to a combination of factors such as mortality, changes in behaviour and the increase of young during the

breeding season. Chaffinch counts were variable across seasons, probably due to flocking. Long-tailed cuckoos were observed at Rolleston Range, and shining cuckoos at Alexander Range, only during the summer months due to the migratory behaviour of both species, which are only present in New Zealand forests between October and January (Heather & Robertson 2005).

Responses of bird populations to the 1080 treatment and mammalian predator levels

Chaffinches were the only species to show a significant decline across treatment sites relative to non-treatment sites, but this did not occur over the same time period at both regions. At Rolleston Range, chaffinch counts increased at both sites over the first post-1080 interval, probably due to an increase in conspicuousness during summer. However, counts decreased only in the treatment site during the second interval. If chaffinches had suffered a decline due to 1080 poisoning, levels would be expected to decrease immediately after the 1080 operation had been conducted. The baits generally only remain on the ground for around one to two weeks after an operation before biodegrading (King et al. 1994), so this decrease during the second interval occurred during a time when 1080 poisoning risk was non-existent. Chaffinches at Alexander Range showed a different pattern, with counts at both sites decreasing immediately after the 1080 operation and then increasing only at the treatment site over the second interval.

It is possible that the decrease in chaffinch levels was due to an indirect delayed effect of the treatment. Silvereyes and grey warblers declined after the eradication of kiore (*Rattus exulans*) on Tiritiri Matangi Island in 1993, whereas bellbirds increased (Graham & Veitch 2002). Innes et al. (2010) suggested that this interaction could perhaps be due to competition for food resources, although this has not been tested. In our study, bellbird, silvereye and tomtit levels were high at the treatment site during both summer seasons, and it is possible that chaffinches were competing with these species for invertebrates. Figure 3.3 (on the following page) shows that the average abundance of 'competitors' per count (bellbirds, silvereyes and tomtits, plus tui at Alexander Range) increased over summer seasons.

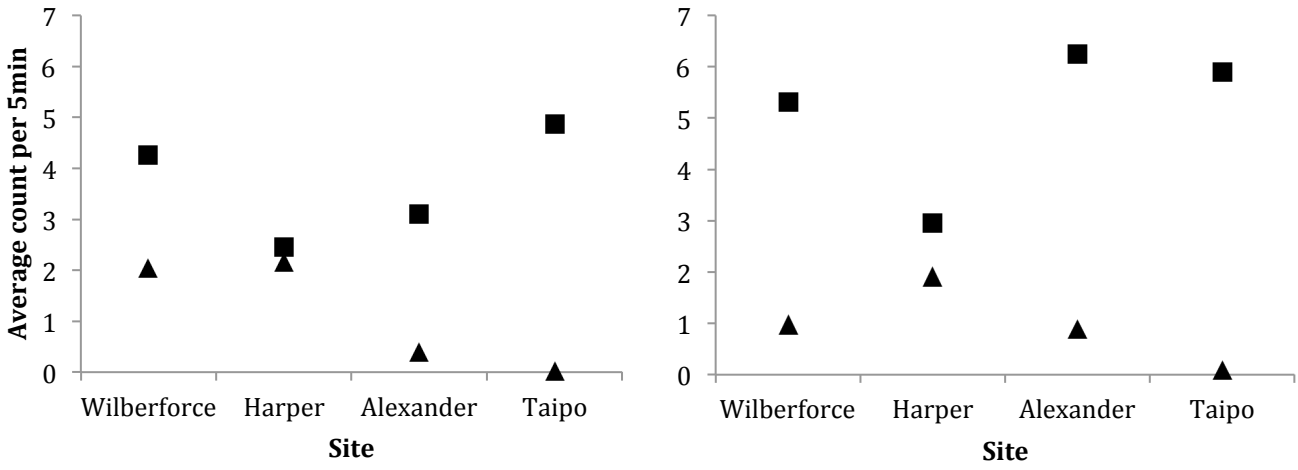

Figure 3.3. Average number of birds per 5min count for chaffinch (triangles) and competitors (squares; combined averages of bellbird, tui, silvereye and tomtit) over summer 2012/13, left, and summer 2013/14, right, at the Rolleston and Alexander ranges.

Recent studies of forest bird populations have found that aerial 1080 operations may provide long term benefits to bird species, as reducing mammalian pest levels often results in increased survival and nesting success of species that are vulnerable to predation (O'Donnell & Hoare 2012, Greene et al. 2013). Removal of these predators can also benefit native species by reducing competition for shared food supplies such as invertebrates, flowers, fruit and leaves (Innes et al. 2010). In many of these studies, the presence of both possums and rodents at high levels before eradication meant that the effects of possum control could not be separated from those of rodent control (O'Donnell & Hoare 2012, Greene et al. 2013). Possums are known predators of native birds and their eggs, and are also thought to impact bird species through forest modification and resource competition, but few studies have been able to quantify these impacts for common forest birds (Sweetapple et al. 2004).

The aerial 1080 operations at the Rolleston and Alexander regions significantly reduced possum levels, but had no effect on detection rates of rats. Out of all four study sites, levels of rats were highest at the Alexander treatment site, with 34% of chew cards marked by rats during winter 2012. There was no significant impact of the 1080 operation on reducing rat levels, with the summer 2013/14 detection rate remaining at 34%. In contrast, rat levels at the non-treatment site at Alexander remained low throughout the study. Mice were detected at low levels throughout the monitoring period at the Alexander region, but were significantly reduced at the Rolleston treatment site immediately following the 1080 operation. By summer 2013/14, mice levels had increased to be higher than pre-1080 levels. Mice are primarily implicated in the decline of native birds by causing stoat population irruptions

when mouse populations increase as a result of beech masting, resulting in higher predation of native birds by stoats (King 1983). No stoats were recorded in tracking tunnels at either region throughout the monitoring period, although they are probably still present at low numbers.

Despite the failure of the 1080 operation to reduce rat levels at the Alexander treatment site, bellbirds and silvereyes showed a significant increase in response to the removal of possums, with counts of both species lower at the treatment site prior to 1080 and increasing at a higher rate than the non-treatment site over the summer seasons. However, there was no significant change in bellbird and silvereye counts at the Rolleston treatment site that wasn't also observed at the non-treatment site, suggesting that increases at Alexander Range may not be solely attributable to the 1080 operation.

Tomtit counts significantly increased at both treatment sites, although the patterns in the non-treatment sites were not consistent across the two regions. Seasonal changes in conspicuousness were observed at both regions, with counts lower during winter.
At the Alexander Range tomtit counts were higher at the non-treatment site throughout the study, but counts increased significantly more at the treatment site over the first and second post-1080 seasons. At the Rolleston range tomtit counts were higher at the treatment site throughout the study, but also continued to increase over both intervals while declining in the non-treatment site over the second interval. The response of tomtits at the Rolleston treatment site is consistent with a highly successful breeding season during 2012/2013. During this period, both possum and mouse detection levels were near zero. Mice have occasionally been recorded eating small eggs and nestlings, but are not often associated with nest predation (King 2005). Tomtit counts at the non-treatment site continued to decrease despite no detection of mice during summer 2013/14, which indicates that predation by mice is not an important limiting factor for tomtit populations at the region. Tomtit counts at the Rolleston treatment site remained stable but did not increase in summer 2013/14, suggesting that the population may have reached carrying capacity following the successful breeding season in summer 2012/13.

The fact that tomtit levels did not significantly decrease provides further evidence towards research finding that cereal baited 1080 operations have no net negative impact on tomtit populations (Westbrooke & Powlesland 2005). Dead tomtits have been found after cereal baited operations (Spurr & Powlesland 1997), but previous reports of high tomtit mortality have largely resulted from carrot baited operations, with Powlesland et al. (2000) reporting a 79% decrease (11 out of 14) in banded North Island tomtits after one such operation in 1997.

The increased breeding success of surviving tomtits, as outlined above, means that 1080 applications could provide an overall net positive benefit.

Measurements of bird density and limitations

The number of five-minute counts conducted, along with a conservative approach to analysis, means it is likely that only large changes in numbers (30-50% or higher) could have been detected for most bird species. Choquenot & Ruscoe (1999) studied the biological and statistical significance of non-target poisoning assessments, and concluded that species with a high resilience (i.e. capacity to recover from poison-related mortality) should be able to sustain higher levels of reduction due to poisoning than species with a low resilience. Therefore for species with a high resilience, such as tomtits and robins, changes in numbers below what I was able to detect are unlikely to negatively affect the populations in the long-term. In this study, tomtits showed an increase at both treatment sites, so there is no concern that a negative treatment effect has gone undetected. For species with a low ability to recover from a 1080-related mortality event, lower levels of reduction could negatively affect the population and therefore smaller changes in numbers would need to be detected. This largely applies to species with low reproductive rates and high predator vulnerability, such as kiwi and kaka, which were not monitored in this study. However, the ability to detect changes to populations below 30% would give a greater certainty that no native species has been negatively affected by the 1080 operation, and may also be able to detect smaller increases in populations that could be attributed to the treatment.

Automatic bird recorders and rare species

As well as five-minute bird counts, recording bird song using automatic bird recorders was trialed during winter 2012 and summer 2012/13. The time consuming process of scoring the recordings meant that this information was not able to be included in my thesis, but addition of the recording data to the five-minute counts would increase sample sizes and the range of effect sizes that are able to be detected, particularly for the winter 2012 season at the Alexander region where only 40 counts were made at the treatment site.

The five-minute bird count method is more suited to common forest bird species than rare species, as power analyses show that a high number of birds per count are needed to detect small changes between samples. Hartley and Greene (2013) suggest that the combination of five-minute counts with other survey methods such as line transect sampling would provide more accurate information for rare birds.

In conclusion, monitoring of common forest bird species over one pre-treatment and two post-treatment seasons showed that no native species suffered a significant decline in response to the 1080 application. Tomtits were the only species to show a significant increase at both treatment sites relative to the non-treatment sites. Levels of one exotic bird species, the chaffinch, significantly declined at both treatment sites relative to the non-treatment sites, possibly in response to an indirect delayed effect of the treatment. Overall, the 1080 application has shown no significant costs to native bird species, and predator removal is providing a net benefit to tomtits.

Chapter 4. Weta monitoring

4.1 Introduction

Weta are large, flightless, nocturnal invertebrates (Orthoptera) that are endemic to New Zealand. There are at least 90 species within two families: true weta, Anostostomatidae, and cave weta, Rhaphidophoridae (Gibbs 1998). The true weta family includes giant weta, tree weta, tusked weta and ground weta. The genus tree weta (*Hemideina*) consists of seven species (Wehi & Hicks 2010). Four species, *H. thoracica, H. trewicki, H. crassidens* and *H. femorata*, are common over a range of both modified and unmodified habitats (Gibbs 1998).

Tree weta are relatively long-lived, with an adult life span of up to three years, and reach maturity at around 18 months of age. All species of tree weta occupy galleries in trees and logs, which provide refuge from predators. They are sexually dimorphic and polygynous, with galleries typically occupied by a dominant male and a harem of females (Kelly 2006). There is considerable variation in the head and jaw size of sexually mature males, which appears to be a result of sexual selection for males to defend females in a gallery (Jamieson et al 2000). Tree weta are thought to be mainly herbivorous, feeding on leaves, fruits and seeds, as well as occasionally scavenging on dead or decaying animals (Fadzly & Burns 2010). Tree weta are particularly vulnerable to mammalian predation due to their large body size (4-40g) and slow movement (Gibbs 1998). Several studies have found that tree weta make up a significant portion of the diet of invasive ship rats, often being their most consumed prey item (Innes 2005, Ruscoe et al 2012). In pre-European New Zealand, weta were subject to predation from a number of native predators, including kaka, weka, harrier hawk, morepork, kiwi, tuatara and the short-tailed bat. However, since the introduction of mammalian pests to the mainland during European colonisation and the decline of many native predators, rodents are now thought to be the main predators of weta (Gibbs 1998, Watts et al 2011).

Sherley et al. (1999) considered that invertebrates could be at risk of 1080 poisoning. Not only has the toxin previously been used as an insecticide, but the main active ingredient of 1080, fluoroacetate, originates from a toxic component of plants which serves as a defence against insect herbivory (Eason et al 2011). Captive tree weta have been observed dying after feeding on 1080 baits in the laboratory, and 12 species of true weta, including *Hemideina* spp, have been reported feeding on carrot and/or cereal based baits (Hutcheson 1989, Spurr & Berben 2004).

Despite this, there is no evidence of wild tree weta populations suffering from a decrease in abundance due to 1080 poisoning, and recent studies suggest that weta populations could benefit from 1080 operations (Powlesland et al. 2005, Ruscoe et al. 2012). Spurr & Powlesland (1997) summarised the results of studies that had monitored invertebrate populations during 1080 operations, and concluded that there was not enough information to determine the impact of 1080 on forest invertebrates. In 1999, Spurr & Drew identified invertebrates, including at least eight species of weta, feeding on non-toxic cereal and carrot based baits, but predicted that 1080 operations were unlikely to have a long-term negative impact on invertebrate populations, as the proportion of individuals feeding on the baits was likely to be small compared to the total numbers of invertebrates in the area.

Powlesland et al (2005) monitored numbers of invertebrates, including *H. thoracica* (Auckland tree weta) found in artificial refuges every second or third month for a year prior and two years following an aerial 1080 operation in Whirinaki Forest Park. Results showed no indication of increased mortality of tree weta in the treatment area. There was also evidence of the 1080 operation benefiting weta populations. For the following two years after the 1080 operation, weta levels increased in the treatment area and not the non-treatment area. This was thought by Powlesland et al. (2005) to have occurred due to the decline in possums, ship rats and mice.

Recently, Ruscoe et al. (2012) investigated the effects of mainland rat control, in the form of aerial 1080 poisoning, on native invertebrates, including several weta species. They hypothesized that the prey items that comprised the largest percentage of invasive rat diets would show the greatest response to rat control. *H. thoracica* was the main component of rat stomach contents, and the abundance of tree weta increased in response to decreases in rat density. How long the potential benefits remain for tree weta populations after an aerial 1080 operation is not yet known, but may be affected by weta longevity, the reinvasion rate of rodents and other mammalian predators, and the frequency of further 1080 applications in the area.

Observing tree weta in the wild can be difficult due to their nocturnal behaviour and daytime preference for galleries that are located in living trees, which can often not be found, or if found can't be accessed without destroying the gallery. Artificial shelters provide easy observation, as weta inside the shelter can be counted and measured with minimal disturbance. Bleakley et al (2006) compared numbers of weta in artificial shelters with numbers estimated from a search of natural cavities in the surrounding area, and found that the number of weta in artificial shelters was likely to reflect the number of weta occupying

natural shelters. Studies using weta shelters as a monitoring method have observed weta beginning to occupy shelters from one to nine months after installation, with occupancy rates increasing over time (Bleakley et al. 2006).

Objectives

This chapter presents the results of weta monitoring. The purpose of the monitoring was to determine how tree weta populations responded to a single aerial 1080 application, using tracking tunnels and weta shelters. The monitoring was carried out over three seasons (one pre-treatment and two post-treatment) at the Rolleston and Alexander ranges. Relative indices of tree weta populations (tracking tunnel rates) were compared between treatment and paired non-treatment sites and over time. Monitoring was also carried out at a nearby non-treatment site, Cass, where weta shelters were established in 2009/2010, in order to compare weta occupancy rates with the shelters installed at the Rolleston range.

Predictions of the response of tree weta to the 1080 operation were as follows:

1) If weta suffer from 1080-related mortality at the treatment sites, this will be seen as a decrease in weta levels immediately post-treatment relative to the non-treatment sites.
2) If weta benefit from the removal of predators, levels will increase over one or both intervals at the treatment sites relative to the non-treatment sites.
3) If rodent levels increase above initial densities by the second interval, weta levels may decline from one year post-treatment.

4.2 Methods

4.2.1 Sites

Tracking tunnels and weta shelters were used to monitor weta populations at Rolleston Range, Alexander Range and Cass. The first two sites have been described in previous chapters. The Cass Mountain Research Area (CMRA) is located in the Cass River basin, approximately 20 kilometres east of the Harper site. The CMRA is owned by the University of Canterbury and is used as a research area for many environmental studies. The area of mountain beech forest containing Reservoir, Middle and Sugarloaf has been subjected to intermittent possum trapping but not rat control, making the site suitable for use as a non-treatment area.

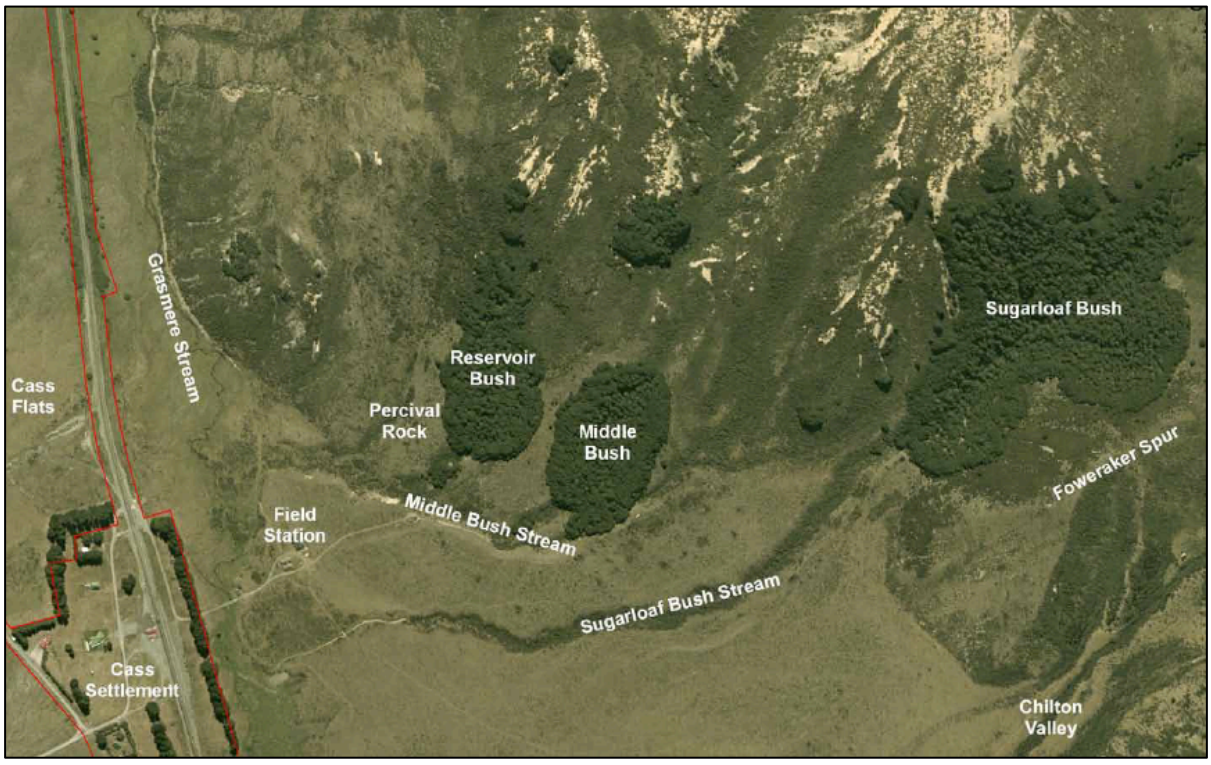

Figure 4.1. Aerial photo showing the sites used for weta shelters (Reservoir Bush, Middle Bush and Sugarloaf Bush) within the Cass Mountain Research Area.

4.2.2 Tracking tunnels

As well as analysing weta footprint data from tracking tunnels at the Rolleston and Alexander sites (see Section 2.2 for full methods), I used tracking tunnels at Cass to obtain relative indices of weta levels. Tracking tunnels were run across three seasons at the Rolleston and Alexander Ranges, but were run only once at Cass during summer 2014 (Table 4.1a). During winter 2012, two sets of tracking tunnels were run. The 'A' set included 50 tunnels per site, placed along five lines of 10 per line and spaced 50m apart. These tunnels were left out for three nights, and baited with 'Erayze' polymer rabbit bait to attract stoats

and rodents. The 'B' set of tunnels included 40 tunnels per site in 4 lines of 10 per line, spaced at 100m and baited with peanut butter, and exposed for two nights). In winter 2012, weather conditions meant that some additional cards from the 'B' set were exposed for 3 and 7 nights at Wilberforce (Table 4.1b).

For comparisons between seasons and sites, the 'B' set tunnels that were exposed for two nights were used for winter 2012, as this method most closely matched the method used post-treatment (Table 4.1a). All pre-treatment data were separately analysed as a comparison of method, to test for the effect of bait type (peanut butter or rabbit), and the number of nights exposed (2-7).

Table 4.1. Dates of tracking tunnel monitoring at the Rolleston and Alexander ranges over three seasons, and Cass during summer 2013/14.
a) standard sets used in full analysis.

	Rolleston		**Alexander**		**Cass**
	Tmt	Non-tmt	Tmt	Non-tmt	
Winter 2012	28-30 June	28-30 June	4-6 July	26-28 July	-
Summer 2012/13	29-30 Nov	4-6 Dec, 22-23 Jan	18-19 Dec	13-14 Dec	-
Summer 2013/14	1-2 Oct 23-24 Jan	14-15 Jan	10-11 Dec	3-4 Dec	25-26 Mar 2014

b) Additional sets from winter 2012, run for various lengths of exposure.

	Rolleston		**Alexander**	
Winter 2012	Tmt	Non-tmt	Tmt	Non-tmt
A set	30 June-3 July	19-22 June	7-10 July	8-11 Aug
B set	16-19, 22-28, 28-30 June	28-30 June, 20-22 July	4-6 July	26-28 July

A total of 30 tracking tunnels were run at Cass, with ten tunnels per forest patch (Reservoir, Middle and Sugarloaf bushes). The tunnels were placed 50 metres apart, and approximately five metres from the exterior of the bush. The tunnels were baited with peanut butter, and exposed for one night.

Tracking tunnel cards from the Rolleston and Alexander Ranges were scored by Josh Van Vianen and Archie Macfarlane, and cards from the Cass area were scored by myself. In summer 2013/14, weta footprints from Rolleston and Alexander were separated into three categories; adult, juvenile and possible juvenile (when it was difficult to distinguish whether the track was from a juvenile weta or a small non-weta invertebrate). For statistical analysis, the adult and juvenile categories were combined, and the 'possible juvenile' category was not included.

4.2.3 Weta shelters

A total of 200 weta shelters were installed between 18-21 February 2013 at Rolleston Range. The shelters, measuring 250mm height x 75mm width from the outside and 20mm deep x 35mm wide from the inside, with an entrance hole of 14mm in diameter, were constructed from untreated pine and are of a single-chamber design (Fig. 4.2). Weta and other invertebrates are able to access the chamber through the small circular hole at the bottom of the shelter. The contents of the chamber can be checked by unscrewing and rotating the front panel.

At each site, 100 shelters were installed along the two lines used for bird count stations and mammal monitoring. The shelters were placed 20m apart, and attached to tree trunks at approximately chest height using coiled stainless steel wire.

Figure 4.2. A weta shelter installed in the treatment area at Rolleston Range, occupied by an adult female tree weta.

I also monitored 72 weta shelters that were first installed between December 2008 and November 2009 at Reservoir Bush, Middle Bush and Sugarloaf Bush within the Cass Mountain Research Area. These shelters measure 225mm height x 75mm width x 30mm deep from the outside, and 130mm x 45mm x 20mm from the inside, with an entrance hole of 14mm in diameter. The shelters are spaced 20m apart in six lines of 12 shelters per line. Data from previous weta monitoring at Cass is available for the years 2010, 2011 and 2012. This

database provides information on shelter occupancy, sex ratios, proximity of shelters to the interior/exterior of the forest, and mark-recapture data.

Weta shelter monitoring

The first weta census was taken on 3 September 2013 at Cass, and 16-17 September (seven months after installation) at the Rolleston region. Each shelter was checked for weta and other invertebrates, and all species present were recorded, along with the shelter line and number. Measurements of tree weta body length (from the tip of head to the rear of the abdomen, excluding the ovipositor, cerci and antennae), right femur length, and (if male) the head length and width, were taken using digital calipers. The sex of weta was recorded, except in the cases of small juveniles where sex could not be determined. Distinguishing features such as missing limbs were also recorded.

4.2.4 Weta mark-recapture

During the first census, each weta found was marked using an insect marking paint kit (Ecroyd Beekeeping Supplies Ltd., Christchurch). Three dots of various colours (yellow, red, blue, green or white) were applied to the right femur to provide each individual with a unique combination, and allow identification of individuals.

Following previous protocol of mark-recapture sampling in the area, the second census at Cass was taken a month after the first census, on 1 October 2013. The same time period between first and second sampling events was not possible for the Rolleston region because of difficulties with site access due to weather conditions; the second census was therefore taken between 14-21 January, four months after the first.

During the second census, the location (shelter number) of each weta found was recorded, as well as the colour combination if the weta had been previously marked. If the weta had no markings, the same sex and size information was obtained as during the first census. Some marked weta showed erosion of one or two dots (particularly the yellow colour), and in these cases the weta could be identified by comparing the remaining dots with the combination recorded in the same shelter during the first census.

4.2.5 Statistical analysis

Tracking tunnel methods were compared using a binomial GLM, with ‘method’ (exposure nights and bait type) as a predictor. Tracking tunnel analyses of total weta footprints (adult and juvenile combined) from Rolleston and Alexander were conducted using the same statistical methods as for mammal tracking tunnel analysis (Chapter 2.2).

Cass and Rolleston summer 2014 tracking tunnels were compared using a binomial GLM, with 'site' as a predictor. Occupancy of interior vs. exterior weta shelters at Cass were compared using a two-sample t-test, assuming unequal variances.

4.3 Results

4.3.1 Exposure nights and bait

As with the mammal tracking tunnel monitoring, analysis for effects of the 1080 operation included only data from the 'two night, peanut butter bait' method for winter 2012. A GLM analysis of the different exposure night/bait combinations used during winter 2012 showed that there was no significant effect of method on weta tracking tunnel rates (p=0.37, Fig. 4.3). This suggests that the means derived from the two and one night peanut butter bait methods used during winter 2012 and the summer seasons are likely to be comparable.

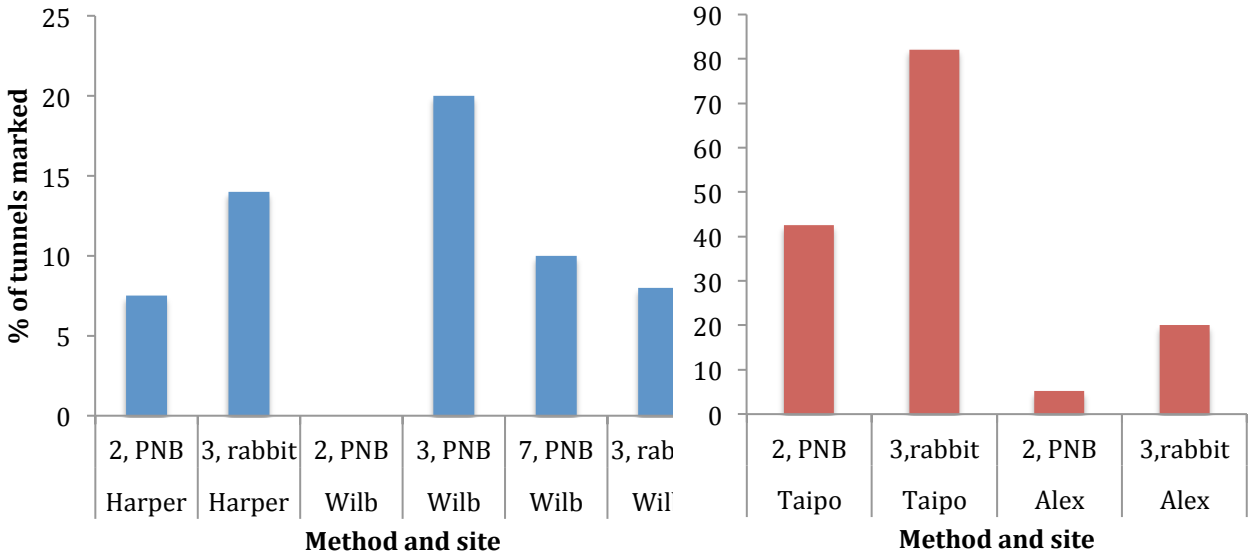

Figure 4.3. The percentage of tracking tunnels marked by weta during winter 2012 at the Rolleston (left) and Alexander (right) Ranges. 'Method' refers to the number of exposure nights and type of bait used (PNB = peanut butter).

4.3.2 Tracking tunnels

Rolleston and Alexander Ranges

Weta footprints were found in tracking tunnels at both the Rolleston and Alexander Ranges (Table 4.2, Fig. 4.4). All weta tracks obtained during summer 2013/14 at both the treatment and non-treatment sites at Rolleston Range were identified as juveniles or possibly juveniles (when difficult to distinguish between juvenile weta or small non-weta invertebrate), whereas most weta tracks at Alexander Range were adult (Table 4.2).

GLM analyses showed significant effects of site and season at both the Rolleston and Alexander regions, but neither region showed a significant season x treatment interaction (Table 4.3). Overall, there was no observed effect of the aerial 1080 operation on total weta levels.

Table 4.2. Tracking tunnel records of weta footprints at a) Rolleston and b) Alexander over three seasons. Method includes number of nights exposed and bait type (PNB = peanut butter). "Total cards" is the number of cards exposed.

a) Rolleston Range

Season	**Winter 2012**	**Winter 2012**	**Summer 12/13**	**Summer 12/13**	**Summer 13/14**	**Summer 13/14**
Treatment	Non	Tmt	Non	Tmt	Non	Tmt
Method	2, PNB	2, PNB	1, PNB	1, PNB	1, PNB	1, PNB
Adult	3	0	21	1	0	0
Juvenile	-	-	-	-	19	0
Possible juvenile	-	-	-	-	9	3
Blank	25	17	4	46	8	11
Total cards	40	40	50	50	50	50

b) Alexander Range

Season	**Winter 2012**	**Winter 2012**	**Summer 12/13**	**Summer 12/13**	**Summer 13/14**	**Summer 13/14**
Treatment	Non	Tmt	Non	Tmt	Non	Tmt
Method	2, PNB	2, PNB	1, PNB	1, PNB	1, PNB	1, PNB
Adult	17	1	35	24	27	7
Juvenile	-	-	-	-	6	4
Possible juvenile	-	-	-	-	1	3
Blank	18	13	6	22	11	15
Total cards	39	19	50	50	50	50

Table 4.3. Summary of significant predictors for Rolleston and Alexander weta (adult and juvenile combined) tunnel data in binomial GLMs (P values for the effects listed, quasibinomial for Alexander). For full details of GLM see Appendix C.

Species	**Season**	**Treatment**	**Season:treatment**
Rolleston weta	**0.002**	**<0.001**	0.112
Alexander weta	**0.043**	**0.001**	0.426

At both regions, weta footprint rates were higher in the non-treatment site throughout the study (Fig. 4.4). Weta footprints at Rolleston remained low at the treatment site, with a 2% detection rate in summer 2012/14 and no detection during winter or summer 2013/14.

At the non-treatment site, levels increased from 8 to 42% over the first interval, and were slightly lower (38%) during summer 2013/14.

At the Alexander region, weta footprints at the treatment site peaked in summer 2012/13 at 48%, before lowering over the second post-1080 interval to 22%. Levels at the non-treatment site also increased from 44% during winter 2012 to 70% during summer 2012/13, and

remained steady during summer 2013/14. The significant effect of season at both regions, with weta observations lower during winter, is likely because tree weta are known to be less active during colder months (Rufaut & Gibbs 2003).

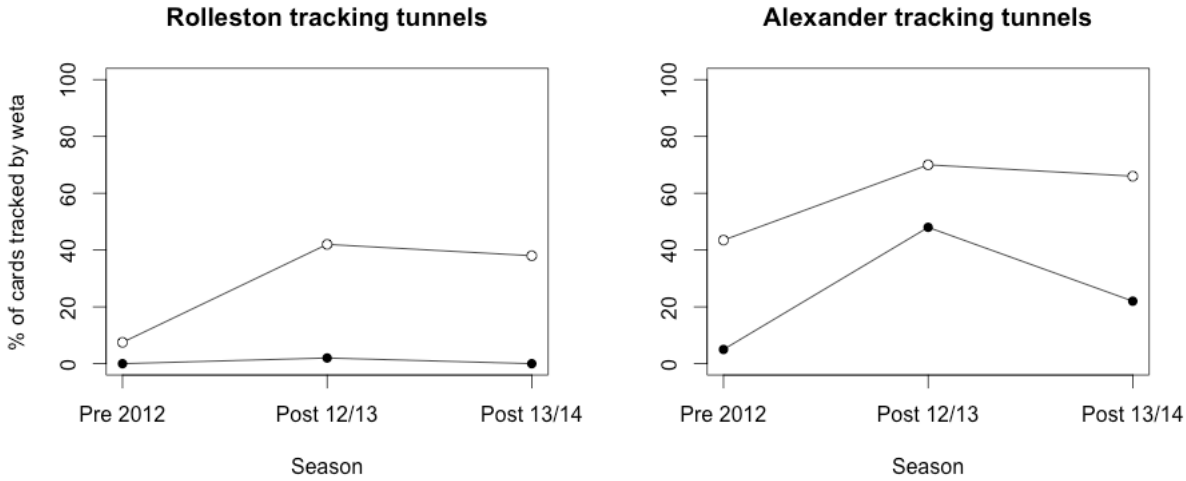

Figure 4.4. The percentage of tracking tunnel cards marked by total weta (adults and juveniles) during monitoring over three seasons at the Rolleston and Alexander Ranges. Black circles represent the treatment sites, and hollow circles represent the non-treatment sites.

Cass tracking tunnels

A total of nine out of 30 tracking tunnel cards (30%) contained weta footprints during tracking tunnel monitoring at the Cass site in March 2014 (Fig. 4.5). Weta tracking rates from Cass were compared with tracking rates from Wilberforce and Harper, which were run in January 2014. GLM analysis (Table 4.4) showed that weta tracking rates were significantly different between the Wilberforce (where no tracks were observed) and the other two sites, but were not significantly different between Harper and Cass.

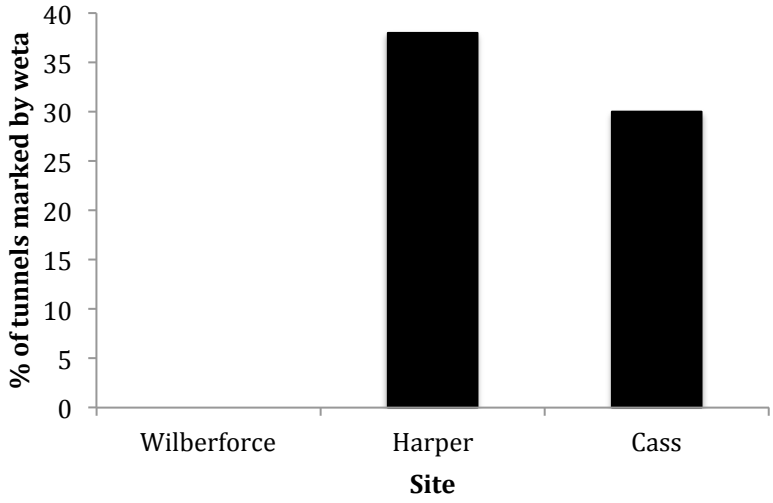

Figure 4.5. The percentage of tracking tunnels marked by total weta (adults and juveniles) at the Wilberforce (treatment) and Harper (non-treatment) sites in the Rolleston region in January 2014, and at Cass in March 2014. No weta tracks were observed at the Wilberforce site.

Table 4.4. Summary of significant predictor (area) for summer 2014 tracking tunnels at Cass and Rolleston in binomial GLM (P values for the effects listed).

Factor	**Df**	**Deviance**	**P**
Area	2	25.139	**<0.001**
Residual	10	16.88	

Coefficients	**Estimate**	**Std. Error**	**Z value**	**P value**
Intercept	-0.8473	0.3984	-2.127	0.03
AreaHarper	0.3577	0.4936	0.725	0.469
AreaWilberforce	-3.0445	1.086	-2.804	**0.005**

4.3.3 Weta shelters

Rolleston

Low numbers of tree weta were found in the Rolleston shelters during both sampling periods (Table 4.5, Fig. 4.6), with an average of 0.025 weta per shelter 6 months after installation in September (5 weta in 200 shelters), and 0.05 per shelter 11 months after installation in January (10 weta in 200 shelters). All were solitary (one per shelter), except for one shelter with a male-female pair found at Harper in January 2014. Although the length of time between sampling attempts meant that mark-recapture information could not be obtained, two shelters at Harper and one shelter at Wilberforce contained one female weta each in both September and January. Body measurements indicate that it is likely the same weta were occupying these shelters during both times.

Other species found in the shelters included cave weta, bush cockroaches, spiders, beetles and slugs.

Table 4.5. Total numbers of tree weta found in shelters at the Wilberforce and Harper sites (100 shelters per site). Shelters were installed in February 2013, and checked during September 2013 and January 2014.

Date	Sept 2013		Jan 2014	
Site	Wilberforce	Harper	Wilberforce	Harper
Male	0	0	1	1
Female	2	1	3	3
Juvenile	1	1	0	2
Total weta	3	2	4	6
Average weta per shelter	0.03	0.02	0.04	0.06

The average number of weta per shelter during January 2014 was low at both the Wilberforce and Harper, with averages of 0.04 and 0.06 respectively (Table 4.5). This shows a much smaller difference between sites than tracking tunnel data from the same time period, where weta detection levels were 38% at Harper, and no tracks were observed at Wilberforce (Fig. 4.4).

Cass weta shelters

The first records available of weta shelter monitoring at Cass were five months after the installation of the D-F line, and 1 year 4 months after the installation of the A-C line (Table 4.6). The average number of weta per shelter was initially low (0.139 per shelter on D-F line five months after installation), but has been steadily increasing to reach 1.19 per shelter overall during the latest count in March 2014 (86 weta in 72 shelters, Fig. 4.6). The Cass shelters were occupied faster than those at the Wilberforce and Harper sites (Fig. 4.6).

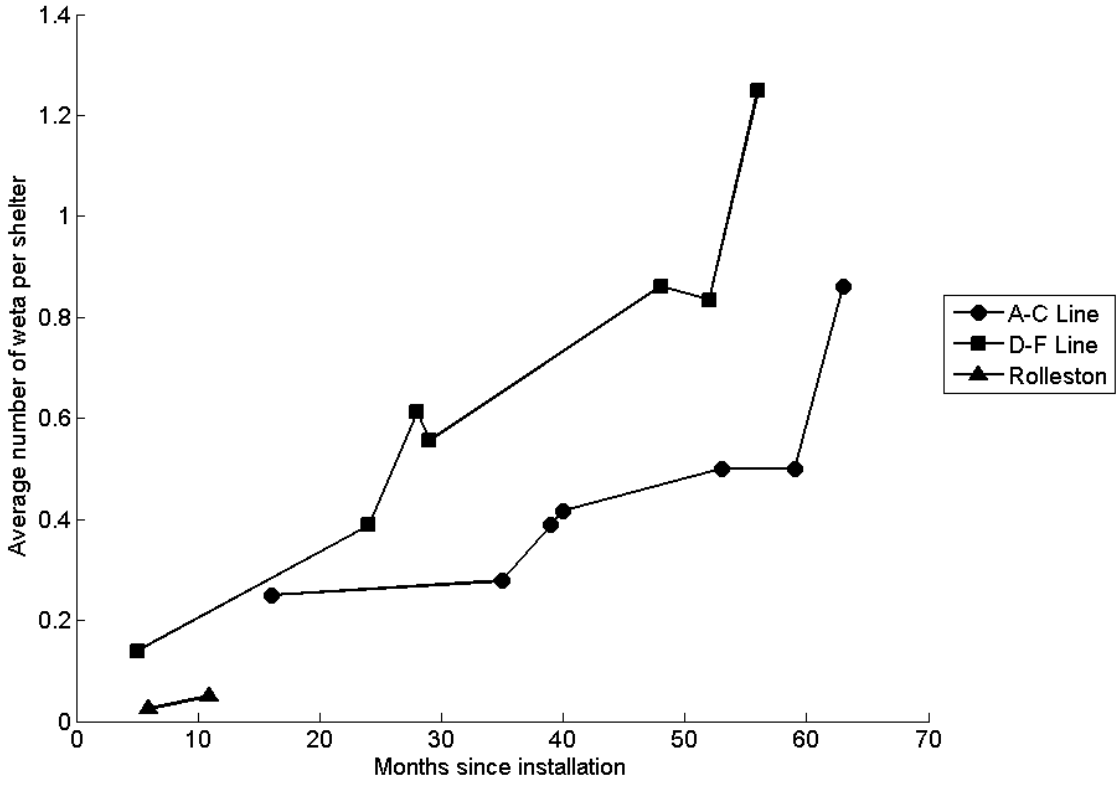

Figure 4.6. Average number of tree weta per shelter during monitoring from April 2010–March 2014 at the Cass and Rolleston regions. The A-C line at Cass was installed in Dec 2008, the D-F line at Cass in Nov 2009, and Rolleston in February 2014.

Table 4.6. Total number of weta recorded in shelters at Cass since installation.
a) A-C line, 36 shelters installed in December 2008.

Date	2010	2011	2012		2013		2014
Time since installation	1.4 years	2.11 years	3.3 years	3.4 years	4.5 years	4.11 years	5.3 years
Male	5	6	7	7	6	7	9
Female	4	4	7	8	12	11	22
Total weta	9	10	14	15	18	18	31

b) D-F line, 36 shelters installed in November 2009.

Date	2010 April	2011 Nov	2012 March	April	2013 Oct	Nov	2014 March
Time since installation	5 months	2 years	2.4 years	2.5 years	4 years	4.4 years	4.8 years
Male	3	6	14	12	15	16	18
Female	2	8	12	8	16	14	37
Total weta	5	14	22	20	31	30	45

Interior and exterior shelters

Of the 72 weta shelters installed at Cass, 36 of these are placed near the forest edge (0-60m from the bush line), and 36 are in the forest interior (over 60m from the forest edge). Since installation, the number of weta found in edge shelters has been significantly higher than the number found in interior shelters (Two sample t-test assuming unequal variances: p=<0.001. Fig. 4.7).

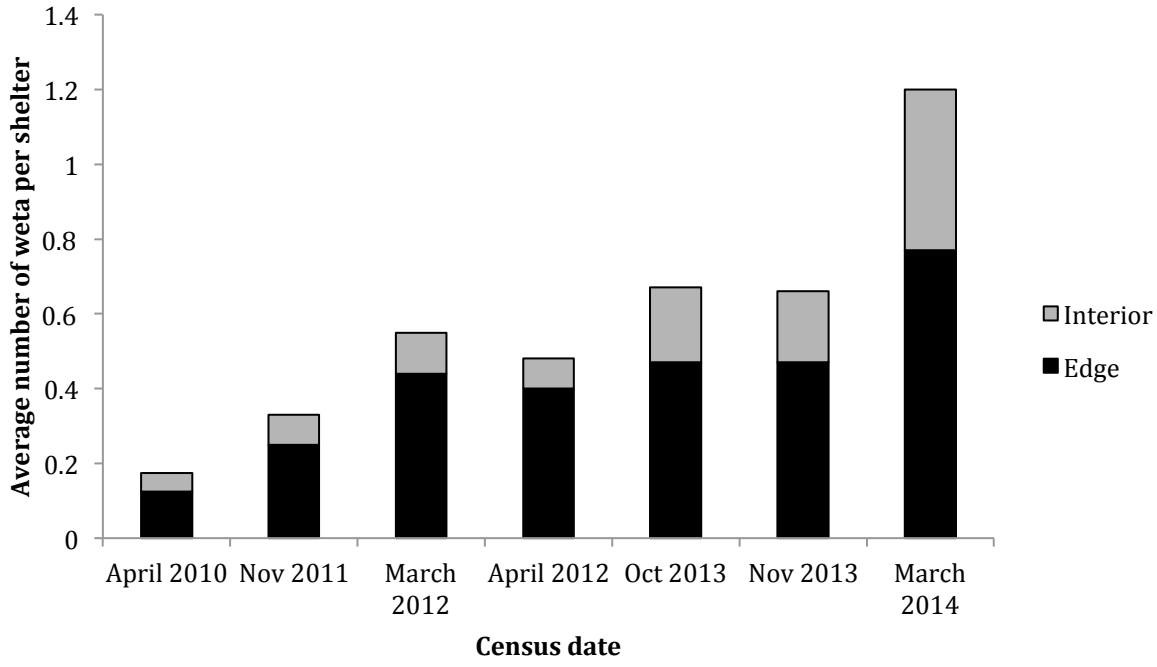

Figure 4.7. The average number of weta found per shelter in interior (grey) or edge (black) shelters during monitoring from April 2010 to March 2014.

4.3.4 Comparing summer 2014 monitoring

Both tracking tunnel and shelter occupancy data were obtained for tree weta at the Cass and Rolleston sites over summer 2014. Shelter occupancy of weta was not similar between the two regions, with an average of 1.19 weta per shelter at Cass, and just 0.04 and 0.06 at Harper and Wilberforce. However, tracking tunnel rates of weta at Cass and Harper in summer 2014 were not significantly different, with rates of 30% and 38% (Table 4.4, Fig. 4.5). Despite the similarity between shelter occupancy rates at Harper and Wilberforce, tracking tunnel rates at the two sites were significantly different, with no weta tracks recorded at the Wilberforce (Tables 4.2a, 4.4).

4.4 Discussion

Effect of the 1080 operation on weta populations

Tracking tunnel rates of weta footprints showed that weta levels at the Rolleston and Alexander Ranges were not significantly affected by the 1080 operation, either immediately after the treatment or over the second interval. This indicates that there were no significant 1080-related mortality events for weta at either treatment site, and that the removal of mammalian predators did not significantly affect weta levels.

In both regions, weta were detected at higher levels at the non-treatment sites both before and after treatment. In particular, tracking tunnel indices at the treatment site at Rolleston were very low throughout the study (0-6%), whereas indices at the non-treatment site ranged from 7.5 to 56%. There was also a seasonal effect for detection rates at the Rolleston range, with lower weta levels observed in winter 2012. This is likely due to seasonal patterns which have been previously observed in weta populations. Tree weta are typically more active on the ground during the summer, when mating occurs and harems are being formed, and temperatures are more favourable for activity in a poikilothermic nocturnal animal (Wehi et al. 2012).

Weta levels in relation to mammalian predator levels

Stoats and rodents are known to be the main predators of tree weta (Gibbs 1998, Watts et al 2011, Ruscoe et al. 2012). Stoats and rats were largely absent at Rolleston Range, but variable levels of mice were detected at both the treatment and non-treatment sites. Mice levels were high as 76% (tracking tunnel rate) at the treatment site during summer 2013/14, and peaked during summer 2012/13 at the non-treatment site with a 52% detection rate. Mice have been found with tree weta remains in their stomach contents (Miller & Miller 1995), but it is unknown how much of a predation risk they pose in comparison with stoats

and rats. Weta levels in the treatment site showed no indication of an increase in response to the eradication of mice following the 1080 operation, and weta levels throughout the study were much higher at the non-treatment site despite the moderate levels of mice detected. This suggests that weta populations at the treatment site may be naturally low, and limited by a factor other than predation.

At Alexander Range, both rats and mice were low in the non-treatment area throughout the study. At the treatment site, rats were detected at levels of up to 34% (chew card rate) during the winter 2012 and summer 2013/14 seasons. Despite the presence of rats, weta levels at the treatment site were much higher than at the Rolleston treatment site.

Occupancy rate of the Rolleston shelters

Occupancy rates of shelters at Rolleston Range were very low, with an average of 0.05 weta per shelter 11 months after installation. Shelters at Cass were also slow to be occupied, taking five years to reach an average of one weta per shelter, although average weta per shelter five months after installation were higher than at Rolleston after six months (0.025 at Rolleston, 0.139 at Cass). Other studies have found tree weta to begin occupying shelters between one and nine months after installation (Trewick & Morgan-Richards 2000, Spurr & Berben 2004, Bleakley et al. 2006).

Given that the tracking tunnel rate for weta has been very low throughout the monitoring period, the weta population at the Wilberforce may be naturally low, explaining the low shelter occupancy. However weta tracking tunnel rates for Cass and Harper in summer 2014 were not significantly different (30% and 38%), suggesting that the low occupancy of Harper shelters is not due to low levels of tree weta.

There are several characteristics of tree weta populations that may contribute to the occupancy rate and usage of artificial shelters. Distribution of weta populations can be patchy, with some research finding that weta may prefer forest edges (Trewick & Morgan-Richards 2000). The shelters at Cass are installed within three small patches of forest, and half of the shelters are considered to be near the forest edge (0-60m from edge). Averages of weta occupancy per shelter are higher for edge shelters than interior ones, suggesting that weta at Cass may prefer forest edges. In contrast, shelters at the Rolleston sites are installed in large areas of forest, with only a few shelters near the forest edge.

It is well established that adult tree weta show fidelity to shelters, often returning to the same one each night, and have small home ranges (Bleakley et al. 2006). Tree weta are also very

slow to disperse, travelling distances of less than 12 metres over one night (Kelly 2006), meaning that dispersal into the areas where the shelters are placed may also be slow. These factors, combined with the number and placement of artificial refuges, are likely to influence occupancy of weta. Previous studies recommend installation of shelters for at least a year prior to monitoring, especially in areas where weta are less common (Bleakley et al. 2006). Powlesland et al. (2005) found a delay of almost two years after shelter installation before adults and harem groups of tree weta were observed. Juveniles of most tree weta species take one to two years to mature, and until that time do not occupy refuges (Powlesland et al. 2005). Tracking tunnel monitoring during summer 2013/14 showed that all weta tracks from Rolleston were likely to be from juveniles, so it is possible that shelter occupancy will increase over the next year with an increase of adults in the population.

Future monitoring of the Rolleston Range is needed to identify any trends in shelter occupancy. As Cass and Rolleston are similar habitats, it is hoped that occupancy of the Rolleston shelters will increase over time at a similar rate to that of the Cass shelters. Tools such as the mark-recapture technique can then be used to gain a more accurate measure of weta populations and their responses to future 1080 operations.

Chapter 5. Discussion

5.1 Overview

The control of introduced mammalian predators in New Zealand forests is crucial for the protection of native species and essential ecosystem services. Possum control in the form of aerial 1080 applications is conducted by TbFree New Zealand to prevent the spread of bovine tuberculosis, and often has the added conservation benefit of temporarily reducing levels of other mammalian predators such as rodents and mustelids. Long-term monitoring provides information on the response of non-target species to 1080 applications and the rate at which mammalian predators recover from the control. This information can help with the management of 1080 applications and allow managers to make decisions based on the benefits that will be provided to native species.

Aerial 1080 operations can have three types of impacts on non-target species: direct poisoning, secondary poisoning, and mesopredator release. Each of these individually could be negative or positive from a conservation point of view, depending on which non-target species are affected. For example, tomtits suffering secondary poisoning would be negative in terms of the conservation of native species, whereas stoats suffering from secondary poisoning would likely benefit native species by reducing predation rates. Given that the target species for most 1080 operations is possums, there is considerable potential for conservation benefits through non-target effects on other exotic species (including rodents, mustelids, and deer). However, none of these effects operate in isolation. For example, tomtits may potentially suffer negative effects from direct toxin ingestion, postive effects from secondary poisoning of stoats, and negative effects from mesopredator release of rats after possum levels are decreased. Effects could be even more complex, for example if exotic chaffinches decrease as a result of increased food competition, after native insectivorous birds increase following secondary poisoning of stoats. This means that the overall outcomes are complex and affected by local conditions, and therefore require careful study.

5.2 Direct poisoning of non-target species

If significant direct poisoning of any non-target species occurred, this would be shown as a decrease in the levels of the affected species observed in the treatment site (but not the paired non-treatment site) immediately following the 1080 application. Aerial 1080 operations are capable of causing high mortality of rodents as a result of direct poisoning (Innes et al. 1995, Powlesland et al. 2000). Mammal monitoring (presented in Chapter 2) showed that mice

were likely the only non-target mammalian pest species to be significantly affected by direct poisoning. Mouse levels significantly decreased at the Rolleston treatment site following the 1080 operation, but recovered to pre-treatment levels by summer 2013/14. Rats were absent from the Rolleston region, as expected for a high altitude mountain beech forest (King 1983, Kelly et al. 2005). Low levels of rats were detected at the Alexander region, but there was no evidence of direct poisoning, as levels were not significantly affected by the treatment. The low level of rats detected meant that the power of our monitoring to detect a large increase (i.e 50%) was also probably low. Nevertheless, tracking tunnel and chew card rates showed that rats didn't increase to reach high absolute numbers, meaning that increased rat predation is unlikely to be a problem for birds.

Native species of birds and weta are thought to be at risk of direct poisoning if they consume 1080 baits (Spurr & Powlesland 1997, Spurr 2000, Veltman & Westbrooke 2011). Although tree weta have been observed to consume 1080 baits in laboratory settings (Hutcheson 1989, Spurr & Berben 2004), monitoring studies have not found decreases in tree weta populations that would indicate large numbers of individuals being directly poisoned following 1080 operations (Spurr & Drew 1999, Powlesland et al. 2005, Ruscoe et al. 2012). Analysis of tree weta monitoring in Chapter 4 showed that weta levels were not affected by direct 1080 poisoning, with no significant decline in weta detection at the treatment site relative to the non-treatment site in the first post-monitoring season.

Many of the bird species observed at the Rolleston and Alexander regions (kereru, rifleman, grey warbler, fantail, tomtit, silvereye, bellbird, tui, blackbird and chaffinch) have had individuals found dead after 1080 operations, although many of these reported deaths occurred before changes were made to operation protocols to deter birds from consuming the baits (Spurr 2000).

Chapter 3 presented the results of bird monitoring, and showed no evidence for a significant decline of any native or exotic bird species due to direct 1080 poisoning. Chaffinch counts decreased at both treatment sites following the 1080 application, but showed different patterns. At one treatment site the decline occurred immediately following the 1080 application, but at the other treatment site declines did not occur until the second post-monitoring period (when 1080 baits would have disintegrated).

Results from this study are consistent with several other long-term monitoring studies (O'Donnell & Hoare 2012, Greene et al. 2013), which suggest that there is no significant negative impact of direct 1080 poisoning on common forest bird populations, as the benefits

of predator control outweigh the costs of possible 1080-related mortality. Species such as robins and tomtits (thought to be particularly at risk of direct poisoning) have been shown to quickly compensate for losses of small numbers of individuals by a post-treatment increase in breeding success, due to lowered nest predation (Powlesland et al. 1999, Powlesland et al. 2000, Greene et al. 2013). Tomtits in this study responded to the 1080 operation with a significant increase in counts at both treatment sites relative to the non-treatment sites. Some other bird species (bellbirds and silvereyes) increased at the Alexander treatment site following the 1080 operation, but this was not matched at the Rolleston treatment site.

5.3 Secondary poisoning of non-target species

Secondary poisoning of carnivores, such as stoats and feral cats, is considered to be a conservation benefit (Gillies & Pierce 1999). However, there is some concern that secondary poisoning may affect insectivorous vertebrates through their consumption of poisoned arthropods. Lloyd & McQueen (2000) measured the 1080 concentrations of arthropods (including tree weta) that were found feeding on baits following an aerial 1080 operation in 1997, and calculated the mean lethal doses for a number of vertebrate insectivores. Lloyd & McQueen concluded that tomtits, robins, hedge sparrows (*Prunella modularis*) and short-tailed bats (*Mystacina tuberculata*) could all be vulnerable to secondary poisoning. Tomtits are the bird species thought to be the most at risk due to their small size and high daily food intake. Lloyd & McQueen (2000) estimated that a tomtit could receive a lethal dose of 1080 by consuming two tree weta (mass 4.7g) that are contaminated with a sub-lethal dose of 12 $\mu g\ g^{-1}$ (a lethal dose for tree weta is estimated to be 91 $\mu g\ g^{-1}$).

If there were 1080-related mortality in native species, determining whether primary or secondary poisoning was responsible is not possible from long-term monitoring studies, but the results of this study indicate that 1080 poisoning did not cause a significant decline in any avian insectivores, as there were no declines in native bird counts at treatment sites relative to non-treatment sites immediately following the 1080 operation. Arthropods that have consumed lethal or sublethal doses of 1080 may be more vulnerable to predation, as the toxin can alter their behaviour (Hutcheson 1989). There was no evidence at the treatment sites of a significant decline in tree weta levels, indicating that insectivores would not have been at high risk of secondary poisoning through the availability of a large number of lethally or sublethally poisoned weta.

5.4 Mesopredator interactions

When only one pest species is targeted for control, predation or competition release can result in the increase of non-target pest species, potentially reducing the potential benefits of the pest control on native biota (Sweetapple & Nugent 2007, King & Powell 2011, Ruscoe et al. 2011). Mesopredator release of rats as a result of stoat control was predicted by simulation modelling (Tompkins & Veltman 2006), and increases of rats may be worse in areas where stoat control is carried out (Dilks et al. 2003). However, Ruscoe et al. (2011) found little impact of stoat control on rat abundance compared to the removal of possums, and suggested that rat populations are more strongly regulated by food availability and competition than by predation, especially as actual rates of stoat predation on rodents are low. Sweetapple & Nugent (2007) also found that rat abundance can increase as a result of possum control in some native New Zealand forests. The mechanism for the increase was thought to be release from food competition, with a 39% overlap in the diets of rats and possums. Although foliage was a major component of possum diets (33%), seeds and fruit made up 52.5% of possum diets and 74% of rat diets (Sweetapple & Nugent 2007).

This competitive release of rats after possum control can have important consequences for non-target species in forests receiving possum control. Tompkins & Veltman (2006) predicted that the presence of possums at high densities could reduce rat abundance by up to 50%, and field monitoring has confirmed that rat abundance indices can more than double following possum control (Sweetapple & Nugent 2007). With pulsed possum control operations such as those carried out for Tb eradication capable of lowering possum levels for three to seven years between control cycles, there is a possibility of native species being subject to increased rat predation between control events.

Rats were detected at low levels at the Alexander region throughout the monitoring period, and there was no evidence of rat levels significantly increasing in response to possum control in either of the post-treatment monitoring seasons. With no other predators (mustelids or cats) recorded at the treatment site and possums at undetectable levels following the 1080 operation, food availability may be the main factor affecting rat density. Summer 2013/14 rat levels at the Alexander treatment site do not appear to be causing a decline in bird or weta levels, with tomtits, bellbirds and silvereyes all significantly increasing across post-treatment seasons. However an increase in food resources, such as the large seedfall that is expected to occur as a result of beech masting during winter 2014, could allow rat populations to reach more damaging levels at both regions.

5.5 Beech masting and predator cycles

In New Zealand beech forests, beech masting occurs every few years due to the irregular flowering and seeding of *Nothofagus* spp (Wardle 1984). During a masting year a huge number of seeds are produced, increasing food supplies and causing irruptions of rodents and stoats (King & Powell 2011). The heavy seedfall allows mice and rats to breed at higher than usual rates from summer through into winter, leading to a short-lived rodent irruption over the next spring and summer. This increase in rodents results in higher survival of stoat offspring, and the number of first-year stoats emerging in December-February is much higher than usual. In habitats where rats are normally not present, such as high altitude mountain beech forests (King 1983, Kelly et al. 2005), the sudden invasion of rats and stoats can have devastating effects on vulnerable native bird species, increasing nest predation on kaka (Dilks et al. 2003), mohua (O'Donnell et al. 1996) and the yellow-crowned parakeet (Elliot et al. 1996). Rodent numbers generally begin to decline during the summer in response to increased stoat predation and diminishing food supplies, which continues the predation pressure of stoats on birds (Kelly et al. 2005).

Beech masting is expected to produce heavy seedfall in *Nothofagus* spp. forests during winter 2014, resulting in the elevation of rodent and stoat levels over spring and summer 2014/15. This masting event has not influenced mammal or bird indices obtained during summer 2014 monitoring, but monitoring over the next summer season (2014/15) will likely show the responses of predator and bird populations to beech masting.

5.6 Conclusions

Monitoring over one pre-treatment and two post-treatment seasons following aerial 1080 applications in two central South Island native forests has found no evidence that native non-target species consistently significantly declined in response to the operation. The 1080 application was beneficial for at least one bird species, with tomtit counts significantly increasing following the treatment. The reduction of possum levels to at or near zero, and the maintenance of this level with ongoing control operations, is likely to continue to provide an overall net benefit to native flora and fauna.

Ongoing monitoring of birds and weta at the Rolleston and Alexander ranges will continue to provide information on the responses of non-target species to an aerial 1080 application. With the next aerial 1080 operation scheduled in 2016, monitoring will also give further insight into the responses of other predators (mice, rats and stoats) to possum control, and the rate of recovery of possums. The beech masting event currently underway during autumn

2014 also provides an opportunity to monitor the interactions between predators, and how possum control may affect these interactions. The use of automatic bird recorders, combined with five minute bird counts, could increase the power of tests used to determine changes in bird counts and could allow smaller changes to be detected. Hopefully the occupancy rate in the Rolleston weta shelters will increase over time, meaning that mark-recapture methods can be used to provide abundance indices as well as sex and class information for tree weta through the post-1080 period.

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Appendices

Appendix A: GLM analysis of mammal abundance

Rolleston chew cards

Mouse (binomial)

Factor	Df	Deviance	P
Season	2	28.679	<0.001
Treatment	1	37.00	<0.001

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-2.4423	0.5213	-4.685	<0.001
season2014sum	-1.4495	1.1367	-1.275	0.2023
Season2013sum	-0.7357	0.8903	-0.826	0.4086
Treatmenttmt	1.4979	0.6091	2.459	0.0139
Season2014sum:treatmenttmt	2.6351	1.2135	2.172	0.0299
Season2013sum:treatmenttmt	-2.2117	1.3828	-1.599	0.1097

Season x treatment	2	12.490	0.002
Residual	24	26.174	

Possum (binomial)

Factor	Df	Deviance	P
Season	2	32.931	<0.001
Treatment	1	156.144	<0.001
Season x treatment	2	24.194	<0.001
Residual	24	30.118	

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-1.5163	0.3681	-4.119	<0.001
Season2014sum	4.2679	0.7001	6.096	<0.001
Season2013sum	2.7820	0.5020	5.541	<0.001
Treatmenttmt	-0.9260	0.6382	-1.451	0.1467
Season2014sum:treatmenttmt	-5.7174	1.3350	-4.283	<0.001
Season2013sum:treatmenttmt	-4.2315	1.2426	-3.405	<0.001

Rolleston tracking tunnels

Mouse (binomial)

Factor	Df	Deviance	P
Season	2	4.950	0.084
Treatment	1	10.024	0.001
Season x treatment	2	105.332	<0.001
Residual	22	39.212	

Coefficients	Estimate	Std. error	Z value	P value
Intercept	0.08004	0.28307	0.283	0.7773
Season2014sum	-3.97186	1.04905	-3.786	<0.001
Season2013sum	-1.17865	0.46202	-2.551	0.0107
Treatmenttmt	-3.97186	1.04905	-3.786	<0.001
Season2014sum:treatmenttmt	9.01636	1.49350	6.037	<0.001
Season2013sum:treatmenttmt	5.37276	1.15592	4.648	<0.001

Possum (binomial)

Factor	Df	Deviance	P
Season	2	7.495	0.023
Treatment	1	18.10	<0.001
Season x treatment	2	2.338	0.311
Residual	22	27.747	

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-1.5163	0.3681	-4.119	<0.001
Season2014sum	0.3637	0.4951	0.734	0.4626
Season2013sum	-2.1472	1.0776	-1.993	0.0463
Treatmenttmt	-2.3755	1.0751	-2.210	0.0271
Season2014sum:treatmenttmt	-0.3637	1.5119	-0.241	0.8099
Season2013sum:treatmenttmt	2.3755	1.7908	1.326	0.1847

Alexander chew cards

Mouse (quasibinomial)

Factor	Df	Deviance	F	P
Season	2	12.197	2.596	0.095
Treatment	1	0.701	0.298	0.589
Season x treatment	2	6.745	1.435	0.258
Residual	24	53.385		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-3.892e+00	1.548e+00	-2.513	0.0191
Season2014sum	2.375e+00	1.648e+00	1.441	0.1624
Season2013sum	-4.619e-15	2.190e+00	0.000	1.0000
Treatmenttmt	2.234e+00	1.657e+00	1.348	0.1904
Season2014sum:treatmenttmt	-2.710e+00	1.874e+00	-1.446	0.1611
Season2013sum:treatmenttmt	-2.234e+00	2.746e+00	-0.813	0.4240

Rat (quasibinomial)

Factor	Df	Deviance	F	P
Season	2	22.642	3.743	0.038
Treatment	1	27.488	9.087	0.006
Season x treatment	2	3.694	0.611	0.551
Residual	24	75.901		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-3.892e+00	1.757e+00	-2.215	0.0365
Season2014sum	1.449e+00	1.977e+00	0.733	0.4706
Season2013sum	1.709e-15	2.485e+00	0.000	1.000
Treatmenttmt	3.229e+00	1.832e+00	1.762	0.0907
Season2014sum:treatmenttmt	-1.449e+00	2.109e+00	-0.687	0.4985
Season2013sum:treatmenttmt	-3.229e+00	3.087e+00	-1.046	0.3060

Possum (quasibinomial)

Factor	Df	Deviance	F	P
Season	2	7.596	1.870	0.176
Treatment	1	30.27	14.903	<0.001
Season x treatment	2	28.186	6.936	0.004
Residual	24	51.936		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-1.3863	0.5039	-2.751	0.0111
Season2014sum	1.7091	0.6486	2.635	0.0145
Season2013sum	0.3403	0.6819	0.499	0.6223
Treatmenttmt	0.2336	0.6904	0.338	0.7380
Season2014sum:treatmenttmt	-4.4482	1.6480	-2.699	0.0125
Season2013sum:treatmenttmt	-3.0795	1.6614	-1.854	0.0761

Alexander tracking tunnels

Mouse (quasibinomial)

Factor	Df	Deviance	F	P
Season	2	3.537	24.275	0.171
Treatment	1	6.224	18.051	0.013
Season x treatment	2	1.334	16.718	0.513
Residual	20	16.718		

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-3.892e+00	1.010e+00	-3.853	<0.001
Season2014sum	-2.064e-15	1.429e+00	0.000	1.000
Season2013sum	2.283e-01	1.430e+00	0.160	0.8732
Treatmenttmt	-3.125e-16	1.429e+00	0.000	1.000
Season2014sum:treatmenttmt	1.899e+00	1.803e+00	1.054	0.2921
Season2013sum:treatmenttmt	1.990e+00	1.861e+00	1.069	0.2849

Rat (quasibinomial)

Factor	Df	Deviance	F	P
Season	2	14.738	3.265	0.059
Treatment	1	10.016	4.437	0.048
Season x treatment	2	2.342	0.519	0.603
Residual	20	43.612		

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-3.8918	1.5176	-2.564	0.0185
Season2014sum	1.1403	1.7617	0.647	0.5248
Season2013sum	1.3795	1.7654	0.781	0.4437
Treatmenttmt	0.7138	1.8652	0.383	0.7060
Season2014sum:treatmenttmt	1.2840	2.1182	0.606	0.5512
Season2013sum:treatmenttmt	-0.3415	2.3566	-0.145	0.8862

Appendix B: GLM analysis of bird abundance

Analysis of 5 minute bird counts: Rolleston

Bellbird (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	18.383	4.301	**0.016**
Treatment	1	129.954	60.81	**<0.001**
Season x treatment	2	3.889	0.91	0.406
Residual	114	288.01		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-0.2937	0.1614	-1.820	0.0714
Season2013sum	0.0359	0.2263	0.159	0.8741
Season2014sum	0.2019	0.1994	1.013	0.3133
Treatmenttmt	0.8515	0.2038	4.178	<0.001
Season2013sum:treatmenttmt	-0.3085	0.2877	-1.073	0.2857
Season2014sum:treatmenttmt	0.0158	0.2509	0.063	0.9498

Brown creeper (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	81.734	6.710	0.002
Treatment	1	290.274	47.66	<0.001
Season x treatment	2	16.333	1.341	0.266
Residual	114	689.75		

Coefficients	Estimate	Std. error	t value	P value
Intercept	0.1358	0.2198	0.618	0.5380
Season2013sum	0.6812	0.2608	2.525	0.0129
Season2014sum	0.6548	0.2539	2.578	0.0112
Treatmenttmt	-2.4258	0.8997	-2.696	0.0081
Season2013sum:treatmenttmt	1.0980	0.9672	1.135	0.2586
Season2014sum:treatmenttmt	1.3265	0.9436	1.406	0.1625

Silvereye (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	152.329	21.552	<0.001
Treatment	1	3.331	0.942	0.334
Season x treatment	2	0.932	0.1319	0.877
Residual	114	372.31		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-0.3829	0.2171	-1.764	0.0803
Season2013sum	0.7387	0.2638	2.800	0.0060
Season2014sum	1.0525	0.2402	4.381	<0.001
Treatmenttmt	0.0389	0.3320	0.117	0.9069
Season2013sum:treatmenttmt	0.1592	0.3912	0.407	0.6847
Season2014sum:treatmenttmt	0.0417	0.3658	0.114	0.9094

Chaffinch (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	262.30	47.771	<0.001
Treatment	1	38.027	13.851	<0.001
Season x treatment	2	25.358	4.618	0.012
Residual	114	335.73		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-0.6061	0.2139	-2.834	0.0054
Season2013sum	1.3821	0.2393	5.777	<0.001
Season2014sum	1.2575	0.2327	5.405	<0.001
Treatmenttmt	-1.2784	0.5239	-2.440	0.0162
Season2013sum:treatmenttmt	1.1906	0.5475	2.174	0.0317
Season2014sum:treatmenttmt	0.6348	0.5515	1.151	0.2521

Grey warbler (Poisson)

Factor	Df	Deviance	P
Season	2	175.76	<0.001
Treatment	1	4.349	0.037
Season x treatment	2	2.616	0.270
Residual	114	133.13	

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-3.3142	0.5000	-6.628	<0.001
Season2013sum	2.7081	0.5164	5.244	<0.001
Season2014sum	3.1213	0.5070	6.156	<0.001
Treatmenttmt	-1.0553	1.1180	-0.944	0.345
Season2013sum:treatmenttmt	1.0452	1.1337	0.922	0.357
Season2014sum:treatmenttmt	0.7136	1.1271	0.633	0.527

Blackbird (Poisson)

Factor	Df	Deviance	P
Season	2	73.34	<0.001
Treatment	1	69.544	<0.001
Season x treatment	2	2.954	0.228
Residual	114	184.62	

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-4.0073	0.7071	-5.667	<0.001
Season2013sum	2.4423	0.7372	3.313	<0.001
Season2014sum	2.7158	0.7220	3.762	<0.001
Treatmenttmt	2.2028	0.7596	2.900	<0.001
Season2013sum:treatmenttmt	-1.0841	0.7975	-1.359	0.1740
Season2014sum:treatmenttmt	-1.1886	0.7801	-1.524	0.1276

Tomtit (Poisson)

Factor	Df	Deviance	P
Season	2	122.50	<0.001
Treatment	1	166.192	<0.001
Season x treatment	2	22.943	<0.001
Residual	114	153.89	<0.001

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-2.3979	0.3162	-7.583	<0.001
Season2013sum	1.0647	0.3667	2.903	0.003
Season2014sum	0.3008	0.3842	0.783	0.434
Treatmenttmt	-0.8729	0.6583	-1.326	0.184
Season2013sum:treatmenttmt	2.3801	0.6901	3.449	<0.001
Season2014sum:treatmenttmt	2.9701	0.6991	4.248	<0.001

Analysis of 5 minute bird counts: Alexander

Bellbird (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	158.67	23.689	<0.001
Treatment	1	4.453	1.329	0.251
Season x treatment	2	32.313	4.824	0.009
Residual	106	359.42		

Coefficients	Estimate	Std. error	t value	P value
Intercept	0.5803	0.1580	3.671	<0.001
Season2013sum	-0.6767	0.2502	-2.704	0.007
Season2014sum	-1.0314	0.2053	-0.502	0.616
Treatmenttmt	-0.3572	0.3032	-1.178	0.241
Season2013sum:treatmenttmt	-0.0389	0.4246	-0.092	0.927
Season2014sum:treatmenttmt	0.7879	0.3422	2.302	0.023

Brown creeper (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	12.045	1.617	0.203
Treatment	1	26.419	7.092	0.008
Season x treatment	2	16.545	2.220	0.113
Residual	106	349.31		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-1.2729	0.4211	-3.022	0.003
Season2013sum	-0.4216	0.6199	-0.680	0.497
Season2014sum	0.0035	0.5357	0.007	0.994
Treatmenttmt	1.1104	0.5357	2.073	0.041
Season2013sum:treatmenttmt	0.1215	0.7407	0.164	0.870
Season2014sum:treatmenttmt	-1.1422	0.6910	-1.653	0.101

Silvereye (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	74.984	10.10	<0.001
Treatment	1	2.257	0.608	0.437
Season x treatment	2	28.88	3.893	0.023
Residual	106	358.04		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-0.3857	0.2697	-1.430	0.1557
Season2013sum	0.1445	0.3478	0.416	0.6786
Season2014sum	0.5386	0.3147	1.711	0.0899
Treatmenttmt	-2.2046	1.1443	-1.927	0.0567
Season2013sum:treatmenttmt	1.8908	1.1906	1.588	0.1152
Season2014sum:treatmenttmt	2.3108	1.1631	1.987	0.0495

Grey warbler (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	10.876	2.690	0.072
Treatment	1	0.085	0.042	0.837
Season x treatment	2	5.288	1.308	0.274
Residual	106	243.32		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-0.2744	0.1883	-1.457	0.148
Season2013sum	0.2113	0.2396	0.881	0.380
Season2014sum	0.0938	0.2355	0.398	0.691
Treatmenttmt	-0.5812	0.3929	-1.479	0.142
Season2013sum:treatmenttmt	0.6808	0.4409	1.544	0.126
Season2014sum:treatmenttmt	0.5878	0.4343	1.353	0.179

Tui (Poisson)

Factor	Df	Deviance	P
Season	2	120.967	<0.001
Treatment	1	30.427	<0.001
Season x treatment	2	5.110	0.078
Residual	106	133.40	

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-4.3175	0.9998	-4.318	<0.001
Season2013sum	3.6037	1.0101	3.568	<0.001
Season2014sum	3.4535	1.0095	3.421	<0.001
Treatmenttmt	0.6286	1.4139	0.445	0.656
Season2013sum:treatmenttmt	-0.2741	1.4242	-0.192	0.847
Season2014sum:treatmenttmt	0.2717	1.4239	0.191	0.848

Tomtit (Poisson)

Factor	Df	Deviance	P
Season	2	143.248	<0.001
Treatment	1	121.025	<0.001
Season x treatment	2	21.806	<0.001
Residual	106	191.77	

Coefficients	Estimate	Std. error	Z value	P value
Intercept	-0.3285	0.1361	-2.414	0.015
Season2013sum	1.2325	0.1505	8.189	<0.001
Season2014sum	1.2948	0.1472	8.797	<0.001
Treatmenttmt	-1.4145	0.4017	-3.521	<0.001
Season2013sum:treatmenttmt	0.3057	0.4206	0.727	0.467
Season2014sum:treatmenttmt	0.9278	0.4101	2.262	0.024

Rifleman (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	30.30	6.452	0.002
Treatment	1	5.021	2.151	0.145
Season x treatment	2	10.457	2.240	0.112
Residual	106	258.50		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-0.0979	0.1853	-0.529	0.598
Season2013sum	-0.9904	0.3242	-3.055	0.002
Season2014sum	-1.2321	0.3276	-3.761	<0.001
Treatmenttmt	-0.2587	0.3431	-0.754	0.452
Season2013sum:treatmenttmt	0.5867	0.4669	1.257	0.212
Season2014sum:treatmenttmt	0.9842	0.4798	2.051	0.043

Chaffinch (quasipoisson)

Factor	Df	Deviance	F	P
Season	2	50.381	363.95	<0.001
Treatment	1	136.883	227.07	<0.001
Season x treatment	2	19.722	207.34	<0.001
Residual	106	207.34		

Coefficients	Estimate	Std. error	t value	P value
Intercept	-0.8518	0.2594	-3.284	0.001
Season2013sum	-2.6346	0.8858	-2.974	0.003
Season2014sum	-1.5461	0.5128	-3.015	0.003
Treatmenttmt	1.1332	0.3285	3.449	<0.001
Season2013sum:treatmenttmt	2.1721	0.9169	2.369	0.019
Season2014sum:treatmenttmt	0.3438	0.5947	0.578	0.564

Appendix C: GLM analysis of weta abundance

Alexander tracking tunnels (quasibinomial)

Factor	Df	Deviance	F	P
Season	2	7.216	3.6307	0.026
Treatment	1	54.805	54.084	<0.001
Season x treatment	2	4.796	2.3982	0.091
Residual	30	90.733		

Rolleston tracking tunnels (binomial)

Factor	Df	Deviance	P
Season	2	59.166	<0.001
Treatment	1	24.285	<0.001
Season x treatment	2	2.307	0.3155
Residual	46	57.616	